UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 2
290 BROADWAY
NEW YORK, NY 10007-1866

February 8, 2012

Ivan Acosta
Chief, Planning Division – Special Projects
Jacksonville District Corps of Engineers
P.O. Box 4970
Jacksonville, FL 32232-0019

Dear Mr. Acosta:

This letter provides our concurrence with your determination that sediments that are proposed to be dredged from areas of the Arecibo Federal Navigation Project are suitable for ocean disposal at the Arecibo Harbor, PR Ocean Dredged Material Disposal Site (ODMDS).

We have reviewed the provided material and concur with your determination that the referenced materials are suitable for disposal at the Arecibo Harbor, PR ODMDS. National policy allows dredged material testing data to be used to make suitability determinations regarding ocean placement for three years. The three year window for the subject concurrence will expire on February 8, 2015. After three years, the Agencies are required to review available information to determine whether changed circumstances (e.g., spills, discharges) might have altered the character of the sediment sufficiently to warrant the retesting of the material. The reevaluation does not, in itself, automatically trigger a requirement for new sampling or testing.

Please ensure that contract specifications address all requirements detailed in the Arecibo Harbor, PR ODMDS Site Monitoring and Management Plan, dated February 1, 2012.

If you have any questions, please contact me at (212) 637-3799.

Sincerely,

Mark Reiss
Dredging Sediment and Oceans Team
Division of Environmental Planning and Protection

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 2
290 BROADWAY
NEW YORK, NY 10007-1866

April 20, 2011

MEMO FOR THE RECORD

SUBJECT: Review of Compliance with the Testing Requirements of 40 CFR 227.6 and 227.27 for the Arecibo Harbor Federal Navigation Project, Arecibo Harbor, Puerto Rico.

FROM: Mark Reiss
Dredging, Sediments and Oceans Team
Division of Environmental Planning and Protection
EPA Region 2

I. SUMMARY

This memorandum provides comprehensive review and analysis of the Arecibo Harbor Federal Navigation Project maintenance sediment test results. This memorandum addresses compliance with the regulatory testing criteria of 40 CFR Sections 227.6 and 227.27, and the requirements set out in Section 228.15(d)(10). These requirements hereinafter are referred to as the "Regulations."

This evaluation confirms that: 1) all tests required under the Regulations were conducted; 2) this project meets the criteria at 40 CFR Section 227.6 for trace contaminants and Section 227.27 for Limiting Permissible Concentration (LPC); and 3) the dredged material is suitable for placement at the Arecibo Harbor, PR, Ocean Dredged Material Disposal Site (AS).

II. PROJECT DESCRIPTION

The proposal is to dredge and place approximately 220,000 cubic yards (cy) of dredged material at the AS. The project encompassed one reach; sediment core samples were taken from three (3) locations to characterize the sediment (see sampling plan (EPA, 2008)).

III. REGULATORY REQUIREMENTS

In order for dredged material to be suitable for placement at the AS, it must conform to the Regulations. The Marine Protection, Research, and Sanctuaries Act (MPRSA) or "The Act" prohibits dumping of materials into the ocean except as authorized by USEPA or, in the case of dredged materials, by the U.S. Army Corps of Engineers (USACE). Section 102 of the Act directs the USEPA to establish and apply criteria for reviewing and evaluating permit applications (33 U.S.C. Section 1412). The USEPA has adopted such criteria in the Regulations. 40 CFR Section 227.6(a) lists constituents that are prohibited from being placed in the ocean unless only present as trace contaminants in material otherwise suitable for dumping (hereinafter referred to as "listed constituents"). Section 227.27 addresses compliance with the LPC. See also, Section 227.13(c).

Section 227.6(b) states that constituents are considered to be present as trace contaminants only when they are present in such forms and amounts that the "dumping of the materials will not cause significant undesirable effects, including the possibility of danger associated with their bioaccumulation in marine organisms." The regulations set forth criteria for determining the potential for significant undesirable effects in Section 227.6(c). In order to be found environmentally acceptable for ocean placement, it must be found that the liquid phase does not contain any of the listed constituents in concentrations that would exceed applicable marine water quality criteria after allowance for initial mixing (Section 227.6(c)(1)). For the suspended particulate phase (Section 227.6(c)(2)) and the solid phase (Section 227.6(c)(3)), bioassay results must not indicate occurrence of significant mortality or significant adverse sublethal effects due to the ocean placement of wastes containing the listed constituents.

Section 227.27 of the regulations addresses the LPC. For the liquid phase, Section 227.27(a) provides that the LPC is that concentration which does not exceed applicable marine water quality criteria after initial mixing, or when there are no applicable marine water criteria, that concentration of material that, after initial mixing, would not exceed 0.01 of a concentration shown to be acutely toxic to appropriate sensitive marine organisms in a bioassay carried out in accordance with procedures approved by USEPA and USACE. For the suspended particulate phase and the solid phase, Section 227.27(b) provides that the LPC is that concentration of material which will not cause unreasonable acute or chronic toxicity or other sublethal adverse effects based on results of bioassays using appropriate sensitive organisms and conducted according to procedures that have been approved by USEPA and USACE, and which will not cause accumulation of toxic materials in the human food chain.

IV. GUIDANCE FOR TESTING AND EVALUATION OF DREDGED MATERIAL

The discussion below describes how the material proposed for placement at the AS, resulting from the maintenance dredging of the Arecibo Harbor Federal Navigation Project was evaluated for compliance with the requirements of 40 CFR 227.6, 227.27.

Testing of the material was conducted following procedures approved by USEPA and USACE, and contained in the joint USEPA/USACE national guidance "Evaluation of Dredged Material Proposed for Ocean Dumping - Testing Manual" (February, 1991) (the "Green Book") (USEPA/USACE, 1991), and the regional implementation manual developed by the USEPA Region 2 and CENAN (USEPA/CENAN, 1992). These test results were analyzed in accordance with the Regulations to ensure that the proposed placement meets the criteria of Part 227.

Applying the USEPA Region 2/CENAN guidance to this project, the material would be Category I if it meets the Part 227 criteria (including the requirements regarding acute toxicity) and:

- bioaccumulation test results do not exceed the regional Matrix levels for cadmium, mercury, total PCBs (clam), and total DDT; and
- bioaccumulation test results do not exceed the Polychlorinated Biphenyl Worm Tissue Criterion of 113 ppb; and
- bioaccumulation test results for the other bioaccumulative chemicals of concern identified in USEPA/CENAN (1992) do not indicate a potential for undesirable effects using conservative assessment techniques.

Sediments that meet this definition are suitable for placement at the AS as they will not cause significant undesirable effects.

V. RESULTS OF EVALUATION OF THE MATERIAL

A. Evaluation of the liquid phase

The liquid phase of the material was evaluated for compliance with Sections 227.6(c)(1) and 227.27(a). There are applicable marine water quality criteria for constituents in the material, including listed constituents, and the applicable marine water quality criteria would not be exceeded after initial mixing. In addition, liquid phase bioassays run as part of the suspended particulate phase on three appropriate sensitive marine organisms, show that after initial mixing (as determined under 40 CFR 227.29(a)(2)), the liquid phase of the material would not exceed a toxicity threshold of 0.01 of a concentration shown to be acutely toxic to appropriate sensitive marine organisms. Accordingly, it is concluded that the liquid phase of the material would be in compliance with 40 CFR 227.6(c)(1) and 227.27(a). The specific test results and technical analysis of the data underlying this conclusion are described and evaluated in Anamar (2010).

B. Evaluation of the suspended particulate phase

The suspended particulate phase of the material was evaluated for compliance with Sections 227.6(c)(2) and 227.27(b). Bioassay testing of the suspended particulate phase of the material has been conducted using three appropriate sensitive marine organisms: inland silversides (*Menidia beryllina*), mysid shrimp (*Americamysis bahia*), and blue mussel (*Mytilus edulis*). That information shows that when placed at the AS and after initial mixing (as determined under 40 CFR 227.29(a)(2)), the suspended particulate phase of this material would not exceed a toxicity threshold of 0.01 of a concentration shown to be acutely toxic in the laboratory bioassays, and

thus would not result in significant mortality. The specific test results and technical analysis of the data underlying this conclusion are described in Anamar (2010). The factor of 0.01 was applied to ensure that there would be no significant adverse sublethal effects. Moreover, the fact that after placement, the suspended particulate phase would only exist in the environment for a short time, means the suspended particulate phase would not cause significant undesirable effects, including the possibility of danger associated with bioaccumulation, since these impacts require long exposure durations (see USEPA, 1994). Accordingly, it is concluded that the suspended phase of the material would be in compliance with 40 CFR 227.6(c)(2) and 227.27(b).

C. Evaluation of the solid phase

The solid phase of the material was evaluated for compliance with Sections 227.6(c)(3) and 227.27(b). This evaluation was made using the results of two specific types of evaluations on the solid phase of the material, one focusing on the acute (10-day) toxicity of the material, and the other focusing on the potential for the material to cause significant adverse effects due to bioaccumulation. Both types of tests used appropriate sensitive benthic marine organisms according to procedures approved by USEPA and the USACE. The following sections address the results of those tests and further analyze compliance with the regulatory criteria of Sections 227.6(c)(3), 227.27(b), and with EPA Region 2 guidance.

1. Solid phase toxicity evaluation

Ten-day toxicity tests were conducted on project materials using mysids (*A. bahia*) and amphipods (*Eohaustorius estuarius*), which are appropriate sensitive benthic marine organisms. These organisms are good predictors of adverse effects to benthic marine communities (see, USEPA, 1996a). The mortality in project sediments did not exceed mortality in the reference sediment by 10% for mysid shrimp by 20% for amphipods. These results show that the solid phase of the material would not cause significant mortality and meets the solid phase toxicity criteria of Sections 227.6 and 227.27.

2. Solid phase bioaccumulation evaluation

Bioaccumulation tests were conducted on the solid phase of the project material for contaminants of concern identified in the project sampling plan (CENAN, 1997) using two appropriate sensitive benthic marine organisms, sand worm (*Nereis virens*) and bent-nosed clam (*Macoma nasuta*). Those compounds with the potential to bioaccumulate (K_{ow} of approximately 4 or greater) or expected to be present in project sediments based upon the location of contaminant inputs and results of previous sediment sampling; and confirmed to be present in the sediments are included on the bioaccumulation testing list. The bioaccumulation test results were used in evaluating the potential impacts of these contaminants in the material. The determination is that the combined results of the toxicity and bioaccumulation tests indicated that the material meets the criteria of Sections 227.6(c)(3) and 227.27(b) of the Regulations, and that the material is suitable for placement at the AS.

USEPA/USACE (1991) describes an approved process of evaluating bioaccumulation potential using comparative analysis of project sediment bioaccumulation to reference sediment

bioaccumulation, FDA Action levels and evaluation of eight additional factors for assessing the significance of bioaccumulation. These factors are:

- number of species in which bioaccumulation from the dredged material is statistically greater than bioaccumulation from the reference material
- number of contaminants for which bioaccumulation from the dredged material is statistically greater than bioaccumulation from the reference material
- magnitude by which bioaccumulation from the dredged material exceeds bioaccumulation from the reference material
- toxicological importance of the contaminants whose bioaccumulation from the dredged material exceeds that from the reference material
- phylogenetic diversity of the species in which bioaccumulation from the dredged material statistically exceeds that from the reference material
- propensity for the contaminants with statistically significant bioaccumulation to biomagnify within aquatic food webs
- magnitude of toxicity and number and phylogenetic diversity of species exhibiting greater mortality in the dredged material than in the reference material
- magnitude by which contaminants whose bioaccumulation from the dredged material exceeds that from the reference material also exceed the concentrations found in comparable species living in the vicinity of the proposed site

In following this guidance, USEPA Region 2 used a framework for evaluating project sediment bioaccumulation results that involves four consecutive evaluations. In the first three evaluations, the project sediment bioaccumulation test results for each compound of concern are sequentially compared to: a) FDA Action levels; b) reference test results; c) Regional Matrix levels; d) general risk-based evaluations.

If bioaccumulation of an individual contaminant from the project sediment does not exceed the FDA levels in step (a) and: the reference test results in step (b) (markings in columns 5 or 7 of Table 1 indicate project test results that were statistically greater than the reference levels for the clam or worm); or the Regional Matrix levels/ PCB Worm Tissue Criterion in step (c) for a particular compound, this indicates that the placement of the material would not result in adverse effects due to that chemical, and there is no need to further evaluate that individual chemical in the next step. *In this project, reference exposures were not conducted therefore all measured residues that were statistically greater than pretest concentration were carried forth in the evaluation as if they were statistically elevated with respect to reference.* General risk-based

evaluations are conducted in step (d) for compounds not resolved in steps (a) to (c). This fourth evaluation (d) uses all the information and results of the individual chemical evaluations (particularly as these results relate to the eight Green Book factors listed above), to evaluate the solid phase of the dredged material as a whole.

The evaluations described above were used for this project and are discussed below in the order considered.

a.) Comparison to FDA Action levels

There are FDA Action levels for seven compounds (aldrin, dieldrin, a-chlordane, heptachlor, heptachlor epoxide, PCBs, and mercury). The source of FDA Action levels is described in USEPA/USACE (1991). Table 1, Column 18, identifies the relevant FDA Action levels.

Exceedance of an FDA Action level results in a conclusion that the placement of the dredged material would result in significant adverse effects. None of the contaminants for which there are FDA Action levels exceed such thresholds in the tissues of organisms exposed to project sediments for 28 days (see also Table 1).

b) Comparison of Bioaccumulation Test Results to Reference Sediment Test Results

Concentrations of contaminants in tissues of organisms exposed for 28 days to project sediments are generally compared to concentrations in tissues of organisms exposed for 28 days to reference sediment. Reference sediment serves as a point of comparison to identify potential effects of contaminants in the dredged material (USEPA/USACE, 1991). In essence, exposing test organisms to this sediment allows for the prediction of contaminant levels that would result in the test organisms were they "in the wild" at the area from which the reference sediment was taken. In this case, reference exposures were not conducted and therefore any measured concentrations in test organism tissues that statistically exceeded the pretest concentration were concluded to be significantly greater than in reference. Throughout this memorandum, statements regarding project sediment having "greater" or "less" bioaccumulation are referring to calculated differences which are statistically significant at the 95% confidence level. To be environmentally conservative, test values which were below detection levels were estimated at very conservative levels for purposes of statistical comparisons (USEPA/CENAN, 1997)

In cases where bioaccumulation levels are statistically greater (at the 95% confidence limit) than in the reference, further evaluation for potential effects is warranted. A statistically significant difference between test and reference bioaccumulation is not itself a quantitative prediction that an impact would occur in the field, nor is it related to any cause and effect. A key to understanding bioaccumulation and potential adverse impacts is that bioaccumulation is a phenomenon and does not necessarily result in an effect. In addition, depending upon the exposure (concentration and duration), bioaccumulation may cause no harm. On the other hand, as exposure and subsequent bioaccumulation increases, the potential for adverse effects increases.

The following text summarizes the test results comparing bioaccumulation from the project sediments to that measured in pretest organism tissues and therefore treated as statistically greater than reference. (Contaminants for which bioaccumulation from the dredged material were concluded to be statistically greater than the reference in the clam and/or the worm are indicated by a mark in columns 5 and/or 7 for that compound in Table 1.)

Metals

- Five (chromium, copper, lead, mercury and nickel) of the nine metals tested were accumulated by clams to levels significantly greater than pretest concentrations. Four (cadmium, copper, lead, silver) of the nine metals tested were accumulated by worms to levels significantly greater than pretest concentrations. Cadmium and mercury are the only metals that are listed constituents in Section 227.6(a). Cadmium bioaccumulated greater in worms exposed to project sediments.

Pesticides

- Of the 15 pesticides (including DDT congeners) tested, 4,4'-DDE, 2,4'-DDD, a-chlordane, and dieldrin were accumulated by clams to levels significantly greater than pretest concentrations (pretest concentrations were below detection or J values, except 4,4'-DDE) and trans-nonachlor was accumulated by worms to levels significantly greater than pretest concentrations.

Industrial Chemicals

- Total PCBs was bioaccumulated to levels significantly greater than pretest concentrations in both the clams (pretest concentrations were below detection or J values) and the worms

PAHs

- Of the 16 PAHs tested, 2 (fluoranthene and pyrene) were accumulated to levels greater than pretest concentrations by clams exposed to project sediments. No PAHs were accumulated to levels greater than pretest concentrations by worms.

For all metals except copper and both species, the magnitude of accumulation is less than 2 times beyond that measured in pretest tissues. Copper was 4-7 times pretest concentrations. For the remaining contaminants accumulated by organisms exposed to project sediment to greater concentrations than the pretest: pesticides and PAHs accumulated to less than two times pretest in clams and worms; total PCBs accumulated to less than 2 times pretest in the worm (clams could not be reliably calculated since pretest was not detected). In such cases, the level of elevation is equated to the level of accumulation beyond reference and the potential for the actual tissue concentration to be related to an effect on the organism or the food chain (including human health) is further evaluated.

c) Comparison to Regional Matrix Levels and Polychlorinated Biphenyl Worm Tissue Criterion

There are regional Matrix levels for four compounds (cadmium, mercury, PCBs (clam only) and total DDT). The source of regional Matrix levels is described in USACE (1981). Table 1, Column 20, identifies the relevant regional Matrix levels. Bioaccumulation results that exceed the regional Matrix level indicate that the sediment is not suitable for placement at the AS under USEPA Region 2 guidance. A Polychlorinated Biphenyl Worm Tissue Criterion of 113 ppb is defined in 40 CFR 228.15(d)(6)(v)(E) of the regulations (see FR 62659). Bioaccumulation results in worm tissue that exceed the worm tissue criterion indicate that the sediment may not be not suitable for placement in the ocean.

DDT, PCBs, mercury and cadmium were detected in the clam and/or worm tissue at concentrations less than the Matrix value. The worm tissue PCB concentration(s) does not exceed the HARS-Specific PCB Worm Tissue Criterion.

i.) Steady State Considerations for Matrix Compounds

When the end point to which the test data is compared potentially represents a steady-state level, rather than a 28-day level, consideration may need to be given to whether the 28-day test results are representative of bioaccumulation levels that could be expected to occur in the field after placement. The literature was reviewed to determine the degree to which the test results reached steady state, as appropriate. The relevance of adjusting project data to steady state for comparison to regional Matrix levels is discussed below.

PCBs

To assess the rate of bioaccumulation of PCBs and other compounds, Rubinstein, *et al.* (1990) and Pruell, *et al.* (1993) exposed three species of organisms, the grass shrimp *Palaemonetes pugio*; the sandworm *Nereis virens*; and the clam *Macoma nasuta*, to sediments collected from the Passaic River, N.J. Sub-samples of the exposed organisms were removed on various days into the study including days 0, 10, 28, 42, 84, and 180. For the clam tissue, the variance in the concentrations on day 28 and day 84 (by which point the maximum concentration had been reached) overlap, thus indicating that the two are not statistically different and the bioaccumulation on day 28 is at or very close to steady-state. Thus, the clam bioaccumulation for the project sediments using 28-day exposures is acceptable for use as steady-state tissue levels, and was below the Matrix level for total PCB and the HARS-Specific PCB Worm Tissue Criterion. For the worm tissue, variances for days 28 and 180 do not overlap, thus indicating that steady-state was probably not reached in 28 days, although the variance in the data makes it difficult to quantify a real difference. However, if the means for days 28 and 180 from Rubinstein *et al.* (1990) are compared (approximately 1,750 ng/g (nanograms per gram or parts per billion, ppb) for 28 days, and 3,000 ng/g for 180 days) this indicates approximately 58% of steady-state would have been reached in 28 days. If on this basis the worm project data are conservatively adjusted upward by even a factor of two to calculate a steady-state tissue concentration, the dredged material tissue concentration is still below the clam Matrix level for total PCB and does not exceed the PCB Worm Tissue Criterion.

Total DDT

With regard to DDT and its metabolites, the degree to which these compounds reached steady-state was also evaluated. Table 1 contains the project test results for the total DDT, which is the sum of the results for DDT and its metabolites (i.e., DDE and DDD). This level is compared to the Matrix level for total DDT. To assess the rate of bioaccumulation of the DDTs and their metabolites, Lee, *et al.,* (1994) exposed the clam *Macoma nasuta*, to sediments collected from the vicinity of the United Heckathorn Superfund site in Richmond California. The study measured tissue residues and uptake kinetics from exposure to pesticide-contaminated sediments. Results of the study indicate that one parent compound, 4,4-DDT, bioaccumulates much more slowly than 2,4-DDT and the DDT metabolites. The results range from approximately 9 percent of steady state after 28 days for 4,4-DDT, to 55 percent of steady state after 28 days for 2,4-DDT. (Lee, et al., 1994) In the project, DDT metabolites were detected and were statistically greater than the pretest in the bioaccumulation test results for both the clam (4,4-DDE, both reaches) and the worm (4,4'-DDD, Reach A). In order to calculate a steady-state tissue concentration, based on the above study a factor of 11 was applied to the project data for 4,4-DDT, a factor of three to the project data for 4,4-DDD, and a factor of two for 2,4-DDT and the remaining DDT metabolites, assuming the detection limit represents the amount present when "not detected." Using these conservative assumptions, the dredged material tissue concentration is below the Matrix level for total DDT in both the worm and the clam.

Cadmium and Mercury

Cadmium and mercury are not regulated in marine organisms as are essential metals, and, thus, no adjustment for steady state is applicable. The Matrix levels for cadmium and mercury, therefore, do not represent "steady state." Bioaccumulation of these metals is affected by many complex factors, and is essentially linear (Dethlefsen, 1978; Giesy, *et al.*, 1980; V-Balogh and Salanka, 1984). Therefore, there are no adjustments that can be made to reproduce "steady state," and so 28-day test results are used to compare to the Matrix levels.

d.) Risk-based evaluations

The potential for impacts due to compounds that produced greater bioaccumulation from project sediments than pretest levels and for which Matrix levels did not exist, was determined using risk-based evaluations. As noted in Table 1 and the previous discussions, PAHs, chromium, copper, and lead fall into that group for the worm and/or clam.

The toxicological significance of this bioaccumulation was evaluated by: i) consideration of steady-state bioaccumulation and food-chain transfer; ii) comparison to background tissue concentrations; iii) consideration of potential ecological effects; and, iv) consideration of potential carcinogenic and non-carcinogenic effects on human health.

i) Consideration of Steady-State Bioaccumulation and Food-Chain Transfer

Bioaccumulation tests were conducted using 28-day exposure of appropriate sensitive benthic marine organisms to sediment. As previously discussed, for bioaccumulation evaluations involving comparisons with "steady-state" tissue concentrations (as opposed to evaluations using other 28-day tissue concentrations such as the comparison to reference sediment), it may be

necessary to understand the extent to which the organism tissue concentration has reached steady-state. Steady-state may be defined operationally as the lack of any significant difference (ANOVA, alpha = 0.05) among tissue residues taken at three consecutive sampling intervals (Lee, *et al.*, 1989). The 28-day test exposure period was selected as appropriate because most chemicals of concern will reach at least 80% of steady-state in benthic marine organisms within that time frame (Boese and Lee, 1992). For the few chemicals that may not meet steady-state tissue concentrations in 28 days, a factor may be used to adjust the data to steady-state when necessary. In order to better use the tissue concentration results of 28-day bioaccumulation exposure tests to assess the risks posed to the environment from the chemicals requiring further evaluation (see discussion above for the identification of such chemicals), consideration was given to the steady-state concentration of these compounds that could occur in the HARS after extended periods of time. In addition, the potential movement of these compounds through the food chain was considered and appropriate trophic transfer factors applied to adjust the data accordingly, as described below.

Metals

In general, metals bioaccumulate more rapidly than organics and 28-day tests are sufficient to evaluate potential effects (see USEPA/USACE, 1991), for example, arsenic (Naqvi, *et al.*, 1990; Riedel, *et al.*, 1987; Oladimeji, *et al.*, 1984).

Trophic transfer of most metals is not sufficient to qualify as biomagnification (Brown and Neff, 1993). The lack of observed biomagnification for such metals as chromium, copper, and lead is the result of incomplete absorption of metals across the gut, rapid excretion, and dilution in muscle, which represents a large part of the total body weight of most marine animals (Fowler, 1982; Suedel *et al.*, 1994). For purposes of conducting the human health and ecological evaluations below, a conservative trophic transfer coefficient equal to one will be used for these non-biomagnifying metals (Suedel *et al.*, 1994 and references cited therein).

PAHs

The time required for a given PAH to attain a steady-state concentration following exposure to bedded sediments (t_{ss}) is determined primarily by the log K_{ow} of the compound in question (McFarland, 1995; Meador, *et al.*, 1995). Meador, *et al.*, (1995) reviewed nine studies that investigated the attainment of steady-state tissue concentrations of PAHs by various marine invertebrates. In each case, tissue concentrations approached steady-state within several days to two weeks after initiating exposure to both low molecular weight PAHs and high molecular weight PAHs. McFarland (1995) estimated the time to steady-state (t_{ss}) for 15 PAHs based on their hydrophobicity. The t_{ss} values ranged from 3.5 to 326 days. The estimated steady-state concentration of the sum total of the 15 PAHs analyzed by McFarland for sediments collected from typical harbor areas revealed that the mean concentration attained after 28-day bioaccumulation tests was approximately 86% of steady-state. McFarland (1995) concluded that 28-day tests are likely to reflect steady-state. However, even using the conservative approach of adjusting the data to calculate steady-state for the individual PAHs in the project based on McFarland (1995) (using a factor of one, two, or three, as indicated) and summing the results, the project data would still fall below the effects levels as discussed below.

With regard to the potential for biomagnification of PAHs, feeding studies show that assimilation rates from ingested food are extremely low, e.g., more than 98% of the target contaminant remained in an undigested form in fish gut 48 hours after feeding squid containing radio-labeled benzo[a]pyrene to young cod (Corner, *et al.,* 1976) and juvenile Atlantic herring (Whittle, *et al.,* 1977) . PAH metabolites are also transferred through the marine food chain; however, they are absorbed even less efficiently than their parent compounds (McElroy and Sisson, 1989; McElroy, *et al.*, 1991). Up to 99% of the PAH compounds taken up by fish are metabolized and excreted into bile, the usual elimination mode, within 24 hours of uptake (Varanasi, *et al.,* 1989). Similar results are described in Brown and Neff (1993) who evaluated various studies describing trophic transfer. The studies cited in Brown and Neff (1993) indicate a trophic transfer rate for BaP from invertebrates to fish of between 0.02 and 0.23 times the concentration in the ingested invertebrates (Corner, *et al.*, 1976, O'Connor, *et al.*, 1988, McElroy, *et al.*, 1991). This was taken into account when assessing the ecological and human health effects of the project material as discussed below.

ii) Comparison of Test Results to Background Tissue Concentrations

Where data regarding tissue levels of organisms living in the open ocean are available ("background levels"), it is useful to compare those levels with the test levels as part of the risk evaluation (Figure 1, Box c). However, this comparison is not, by itself, definitive. When bioaccumulation in organisms exposed to project sediments is not greater than tissue concentrations in organisms from the vicinity of the remediation site (the background levels), this means that placement of the material would not result in bioaccumulation above existing ambient levels in the general area and thus does not have a potential to cause undesirable effects. When bioaccumulation in organisms exposed to project sediments is greater than these levels, it may or may not be predictive of adverse effects (e.g., it may reflect extremely low "background" levels). Depending on the exposure (concentration and duration), bioaccumulation may cause no harm. However, as exposure increases, the potential for adverse effects increases.

Organisms collected from a broad area of the sea floor in the New York Bight have been collected and analyzed for tissue concentration for bioaccumulative contaminants of concern (Charles and Muramoto, 1990; USACE, 1994; USEPA, 1996f; USEPA, 1997b). These field-generated bioaccumulation results provide a measure of the tissue residues for organisms living in the ocean. Table 1, Columns 16 and 17 summarize the most recent background data. For clam background, data were collected only for the following constituents: all PAHs, aldrin, two DDT compounds, PCBs, and seven of the nine metals analyzed. Where background values exist, none of the compounds accumulated by clams or worms to levels above pretest exceeded background levels in this project, except copper in worms which accumulated to 6 times background levels.

iii) Consideration of Potential Ecological Effects

A review of scientific information was also done to further evaluate the test results with respect to potential ecological impacts for the chemicals requiring further evaluation (above reference and for which there is no Matrix level or dioxin value).

Metals

The potential for ecological effects from the bioaccumulation of chromium, copper, lead, nickel, and silver was evaluated by comparing to its corresponding Water Quality Criterion Tissue Level (WQCTL). The WQCTL is calculated by multiplying the Clean Water Act Section 304(a)(1) Federal water quality criterion chronic value (CV) for the chemical by the empirically determined bioconcentration factor (BCF) for the chemical for a representative marine organism (Lee, *et al.*, 1989). A BCF is the ratio of the concentration of a contaminant in an organism to the concentration of the contaminant in water. Thus, the WQCTL represents the tissue concentration that would be expected in an organism exposed to water containing the chemical at the CV concentration. This level is set to protect 95% of all tested organisms included in the water quality criterion database, thus representing a conservative level of protection (USEPA, 1985b). Table 1 lists the calculated WQCTLs. Sources of CVs and BCFs are USEPA ambient water quality criteria documents (USEPA 1980b, 1980c, 1980d, 1980e, 1980f, 1984a, 1984b, 1985a, 1985c, 1986, 1987b and 1992a) and Calabrese (1984)(for silver). Calculations are shown in attachment A. None of the WQCTLs were exceeded except copper in worms which exceeded the WQCTL by a factor of less than two. Therefore, these bioaccumulation test results do not indicate a potential for undesirable ecological effects.

PAHs

For PAHs, a more definitive method is available for evaluating the potential ecological effects. This method makes use of a direct comparison of total PAH tissue residues and the Critical Body Residue (CBR). This approach is supported by a review of the scientific literature. The CBR approach described by McCarty (1991) was used to evaluate the potential impacts of total PAHs accumulated in the dredged material bioaccumulation test organisms. CBRs are concentrations of chemical residues in organisms which elicit a deleterious biological response associated with narcosis, which is the primary non-cancer effect of PAHs. Narcotic responses measured can be acute (e.g., immobilization or death) or chronic endpoints (e.g., reduced reproduction, fecundity or growth). CBRs are represented as the ratio of the mass of toxicant to the mass of the organism, such as millimoles or micrograms of toxicant per kilogram (mmole or ug/kg) of organism. For the narcosis endpoint, each molecule of individual PAH congeners is generally equipotent, thus the total PAH concentration is compared to the CBR. For example, a 400 ppb dose of naphthalene would elicit a similar toxicity response as 400 ppb of fluorene; if both chemicals are present together at these concentrations, then the dose would equal 800 ppb (see Appendix for Table 1).

As shown in Table 1, total PAH levels in tissues from the dredged material bioaccumulation test were below levels at which chronic adverse effects might be expected from a narcotic mode of action in sensitive aquatic organisms (i.e., fish) as estimated by the CBR.

Effects of Mutagenic, Carcinogenic and Teratogenic PAHs. Applying the uncertainty factor (UF) of 10 and a trophic transfer factor of 0.1 described in the Appendix for Table 1, to the no-effects level for BaP calculated from Hannah, *et al.* (1982), as discussed in the Appendix for Table 1 (8,021 ppb) results in a no-effect level for BaP of approximately 8,000 ppb in benthic tissue,

which is considerably greater than the highest tissue concentration of BaP found in the project bioaccumulation test results (approx. 0.15 ppb). Even when applying the more conservative steady-state factors for BaP and the other carcinogenic PAHs derived from McFarland (1995), as identified above, the calculated concentrations (0.3 ppb for BaP only and 0.48 ppb for total BaP equivalents) are still below the no-effects level; the project tissue concentrations would still be below this no-effect level if the higher trophic transfer factor (0.23) reported by McElroy, *et al.* (1991) was used. Therefore, the most relevant aquatic effects information reviewed indicates that the highest tissue levels accumulated in the dredged material bioaccumulation tests are below the no-effect level.

Another study that was reviewed considered the carcinogenicity of BaP in rainbow trout resulting from embryo microinjection (Black, *et al.*, 1988). A statistically significant number of liver neoplasms was found at a concentration of approximately 200,000 ppb, with non-significant effects at up to one half that concentration. Therefore, using the above across-species UF of 10 and trophic transfer factor of 0.1 results in an aquatic no-effect level of 100,000 ppb. Since this is several orders of magnitude above the highest tissue concentration of BaP for this project, as described above (and even the highest BaP-equivalent levels for human health, as discussed above), this provides additional support for a finding that the test results do not indicate a potential for undesirable effects to the marine environment due to mutagenic, carcinogenic or teratogenic contaminants.

Hall and Oris (1991) reported on experiments that exposed fathead minnows to anthracene during long-term exposures and observed adverse effects on reproduction. The paper reported that a concentration of anthracene in the tissue of the egg in the range of 3,750 to 8,000 ppb resulted in no significant effects on egg hatching or survivorship. Using the same approach for accounting for species-to-species uncertainty and food chain transfer described above and in the Appendix for Table 1, yields a conservative benthic tissue level of 3,750 ppb. Anthracene tissue concentrations from the project bioaccumulation tests are well below this level.

iv) Consideration of Potential Carcinogenic and Non-carcinogenic Effects on Human Health

Human health effects screening levels were developed for those chemicals requiring further evaluation with risk-based methods using conservative estimates of exposure to assess whether these contaminants would accumulate to levels in fish and shellfish that could lead to significant adverse effects to humans. The approach assessed consumption of fish and shellfish to derive conservative estimates of contaminant concentrations in benthic tissue protective of human health using USEPA standard risk-assessment assumptions and the process described in the Appendix for Table 1. Table 1, Column 14 lists conservative human cancer protection levels in benthic organisms for the chemicals which are known or suspected carcinogens that would lead to a human cancer risk level of 10^{-4}. When the bioaccumulation test results for those chemicals are adjusted for steady-state (as previously described), the results are below the human cancer protection levels in Table 1.

Since the analysis used conservative methods, the result represents conservative estimates of risk, or what are in effect plausible upper-bound estimates. Thus, the true risk is highly unlikely

to be greater than estimated and could be much lower. None of the human health cancer protection levels were exceeded in the bioaccumulation test results.

The potential for non-cancer impacts can be expressed as a hazard quotient (HQ), which is the ratio of the average daily intake divided by the toxicological reference dose for the chemical. If the HQ is less than unity (i.e., 1), an adverse noncarcinogenic effect is highly unlikely to occur. If the HQ exceeds unity, an adverse health impact may occur. The higher the HQ, the more likely that an adverse noncarcinogenic effect will occur as a result of exposure to the contaminant in the dredged material after placement. Table 1, Column 15 includes the noncancer protection levels in benthic organisms for the chemicals requiring further analysis that are known to cause, or suspected of causing, non-carcinogenic effects, that would result in a human HQ equal to unity. Those numbers were derived using the conservative assumptions and source materials described in the Appendix for Table 1. The concentrations of the chemicals requiring further evaluation were below the non-cancer protection level.

e.) Evaluation of Solid Phase Bioaccumulation Results for Dredged Material as a Whole

The evaluation of the testing results performed above indicates that the material does not have a potential to cause undesirable effects to aquatic marine biota due to chronic adverse effects including such effects due to mutagenic, carcinogenic, or teratogenic contaminants, or to human health due to cancer or non-cancer effects from the individual contaminants. That evaluation includes the information relevant to the eight factors identified in the Green Book for assessing bioaccumulation test results (USEPA/USACE, 1991). As a final and additional step in the evaluative process, however, it is appropriate to go beyond assessing the individual test results in order to look at the results as a whole so as to provide an opportunity for an integrated assessment of the individual test results (Figure 1, Box d). For example, if a number of the individual bioaccumulation test results were only marginally at or below the relevant levels of concern, it is appropriate to consider this and the other relevant factors to evaluate whether, taken as a whole, the material is unsuitable for placement at the HARS, even though no single individual test result would indicate that outcome.

As indicated above, the following chemicals of concern were bioaccumulated above pretest for the clam and/or the worm: copper, lead, and total PCBs bioaccumulated in both the clam and worm, chromium, mercury, nickel, dieldrin, a-chlordane, two individual PAHs, and two DDT metabolites accumulated only in the clam. Silver, cadmium and trans-nonachlor accumulated only in the worm. In the case of those contaminants with test results exceeding pretest, and which have regional Matrix levels or other decision points, criteria, all were below the relevant Category I value. For the non-Matrix contaminants with test results that exceeded pretest levels, except for copper in worms, none bioaccumulated from project sediments to greater than background levels. Although some of the contaminants that bioaccumulated in the tests can be toxicologically important, in no case did they accumulate to toxicologically important concentrations, even when conservative assumptions were used to evaluate the test results exceeding reference, as described above. All contaminants, except copper in worms, exhibited bioaccumulation test results above pretest which were all below the acceptable human health risk range and acceptable aquatic effects range using conservative approaches and analyses as

described above to evaluate those test results. Copper marginally exceeded the ecological screening level. Thus, an evaluation of the solid phase bioaccumulation test results for the dredged material as a whole considering the factors in the Green Book (Figure 1, Box d) would not indicate a different outcome than that shown by the individual test results themselves; i.e., that the material does not have the potential to cause undesirable effects due to bioaccumulation.

Taking into account all of the above information, it is determined that this material will not cause undesirable effects due to bioaccumulation as a result of the presence of individual chemicals or of the solid phase of the dredged material as a whole. Therefore, it is concluded that the solid phase of the material proposed for placement at AS meets the requirements of 40 CFR §227.6(c)(3), 227.27(b).

VI. OVERALL CONCLUSION ON THE PROPOSED PROJECT

Based upon this review of the results of testing of the sediments proposed for dredging and ocean placement from Arecibo Harbor Federal Navigation Project, the material meets the criteria for acceptability for ocean placement as described in Sections 227.6, and 227.27 of the Regulations, and is suitable for placement at the AS.

FOOTNOTES FOR TABLE 1:

*: Carcinogenic PAHs.

#: Levels represent the conservative level of protection for the sum of the related compounds and their metabolites.

na: Not Available

1. A "X" in this column indicates that the analyte concentration in the test sediment is statistically greater than that of the reference sediment. Means and statistical comparisons were determined using conservative estimates of concentrations for analytes that were below the detection limit (USEPA/CENAN, 1997).

2. Conversion factors from 28-day bioaccumulation results to steady state are obtained from the following sources: for PAH's: from McFarland, 1995; for Aldrin, Dieldrin, Chlordane, DDT, DDD, and DDE: from Lee and Lincroft, et al, 1994; for PCBs: from Pruell, *et al.*, 1993, and Rubinstein, *et al.*; for 1,4-Dichlorobenzene: from de Bruijn, *et al.*; for Endosulfan I, Endosulfan II, Endosulfan Sulfate, Heptachlor and trans nonachlor: from Syracuse Research Corporation, 1996, and McFarland, 1995; for Heptachlor Epoxide: from Veith, *et al.*, 1979.

3. PAH TEFs taken from: USEPA. 1993; Dioxin TEFs taken from: USEPA. 1989.

4. Toxic equivalence for the carcinogenic PAHs are from USEPA (1993).

5. This value represents the 10^{-4} cancer risk level for the carcinogenic PAHs. The total concentration of carcinogenic PAHs is expressed in BaP equivalents (see discussion in the text of the memo).

6. Cancer risk factor or reference dose are not assigned by USEPA in IRIS (USEPA, 1995).

7. FDA limits are from the USEPA/USACE, 1991.

8. This value represents the benthic level expected to result in a no-effect level for possible mutagenic and teratogenic effects in fish from exposure to BaP, which is the most toxic PAH.

9. This value represents the non-specific narcosis effects level (see discussion in Appendix). This value is compared to the sum of all PAHs measured.

10. Calculations are included in the appendix to Table 1.

11. Means of five tissue replicates calculated using conservative estimates where analytes were not detected (USEPA/CENAN, 1992); "U" indicates that all five replicates were not detected.

12. Chemicals for which the bioaccumulation from the dredged material was greater than the reference but less than the Matrix level are indicated by bolding the Matrix level in Column 20.

13. Levels are based on the Regional Dioxin Values.

14. Level is the sum of all dioxin congeners other than 2,3,7,8-TCDD.

15. For this PAH, the no-effect level for possible mutagenic and teratogenic effects in fish is estimated from exposure to BaP, which is the most toxic PAH.

16. Cadmium and mercury do not obey steady state kinetics, therefore, no adjustment is made (see discussion in the text of the memo).

17. Cancer and non-cancer protection levels, based on inorganic arsenic as contained in EPA's IRIS database, are not appropriate for evaluating the potential human health impacts of arsenic bioaccumulation from dredged material, and therefore, are not included in Table 1 (see discussion in Appendix to Table 1).

VII. REFERENCES

Abel, , P.D. and V. Axiak. 1991. Ecotoxicology and the marine environment. Ellis Horwood, New York, pp. 269.

Abernathy, C.O. and E.V. Ohanian. 1992. Non-carcinogenic effects of inorganic arsenic. Environ. Geochem. Health 14: 35

Anamar. 2010. Arecibo Harbor 103 Sediment Evaluation, Arecibo, Puerto Rico. Final Report. Prepared for U.S. Army Corps of Engineers, Jacksonville District. Dated August 2010.

Baumann, P.C., W.D. Smith, W.K. Parland. 1987. Tumor Frequencies and Contaminant Concentrations in Brown Bullheads from an Industrialized River and a Recreational Lake. *Transactions of the American Fisheries Society* 116:79-86.

Black, JB, A.E. Maccubbin, and C.J. Johnston. 1988. Carcinogenicity of benzo(a)pyrene in rainbow trout resulting from embryo micro injection. *Aqua Tox.* 13, 297-308.

Breteler, R. (ed.). 1984. Chemical Pollution of the Hudson-Raritan Estuary. NOAA Technical Memorandum NOS OMA 7. National Oceanic and Atmospheric Administration, National Ocean Service. Rockville, Md.

Brown B., J. Neff. 1993. Bioavailability of Sediment-Bound Contaminants to Marine Organisms. Report #PNL-8761 UC-000 by Battelle/Marine Sciences Laboratory prepared for the National Ocean Pollution Program Office, NOAA.

Bryan, G.W. and W.J. Langston. 1992. Bioavailability, accumulation and effects of heavy metals in sediments with special reference to United Kingdom estuaries: A review. *Environmental Pollution* 76: 89-131.

Calabrese, A. 1984. "Effects of Long Term Exposure to Silver and Copper on Growth, Bioaccumulation and Histopathology in the Blue Mussel (Mytilus edulis)." *Mar. Envir. Res.* 1, 253-274.

Call, D.J., L.T. Brooke, M.L. Knuth, S.H. Poirler, and M.D. Hoglund. 1985. Fish subchronic toxicity prediction model from industrial organic chemicals that produce narcosis. *Environ. Tox. Chem.* 4, 335-341.

Charles, JB and J. Muramoto. 1990. Assessment of Contaminants in Sediment and Biota at the Mud Dump Site, New York Bight. Report No. SAIC-91/7608&256 by Science Applications International Corp. (SAIC) for USEPA - Region 2.

Corner, E.D.S., R.P. Harris, K.J. Whittle, and P.R. Mackie. (1976). Hydrocarbons in marine zooplankton and fish. *In*: Effects of Pollutants on Aquatic Organisms, Lockwood APM (ed), pp. 71- 106. Cambridge University Press, Cambridge, England.

de Bruijn, J., Busser, F., Seinen, W., and Hermens, J. 1989. Determination of octanol/water partition coefficients for hydrophobic organic chemicals with the "slow stirring" method. *Environ. Toxicol. Chem.*, 8:499-512.

Dethlefsen, V. 1978. Uptake, retention, and loss of cadmium by brown shrimp. 1978. *Meeresforschung*, 26:137 (reported in Giesy *et al.* 1980).

EPA. 2008. Letter from Mark Reiss to Ivan Acosta, dated June 16, 2008. EPA Region 2, Division of Environmental Planning and Protection.

Feroz, M. And M.A.Q. Khan, 1979. Fate of ^{14}C-cis-chlordane in goldfish, *Casassius auratus* (L.). *Bulletin of Environmental Contamination and Toxicology* 23:64-69.

Finger, E.F., E.F. Little, M.G. Henry, J.F. Fairchild and T.P. Boyle. 1985. Comparison of laboratory and field assessment of fluorene - Part 1: effects of fluorene on survival, growth, reproduction, and behavior of aquatic organisms in laboratory tests. *In:* Validation and Predictability of Laboratory Methods for Assessing the Fate and Effects of Contaminants in Aquatic Ecosystems. ASTM STP865, Philadelphia, pp. 120-133.

Fowler, S.W. 1982. Biological transfer and transport processes. *In:* Pollutant Transfer and Transport in the Sea. Vol. II, ed. G. Kullengerg, pp. 1-65. CRC Press, Boca Raton, Florida.

Giesy, J.P., Bowling, J.W., and Kania, H.J. 1980. Cadmium and zinc accumulation and elimination by freshwater crayfish. *Arch. Environm. Contam. Toxicol.*, 9:683-697.

Gobas, F. 1993. A Model for predicting the bioaccumulation of hydrophobic organic chemicals in aquatic food webs: application to Lake Ontario. *Ecological Modeling.* 69,1-17.

Hall, A.T. and J.T. Oris. 1991. Anthracene reduces reproductive potential and is maternally transferred during long-term exposure in fathead minnows. *Aquatic Toxicol.* 19,249-264.

Hannah, JB, J.E. Hose, M.L. Landolt, B.S. Miller, S.P. Felton, and W.T. Iwaoka. 1982. Benzo(a)pyrene-induced morphologic and developmental abnormalities in rainbow trout. *Arch. Environ. Contam. Toxicol.* 11,167-171.

Holcombe, G.W., G.L. Phipps, and J.T. Fiandt. 1983. Toxicity of selected priority pollutants to various aquatic organisms. *Ecotoxicol. Environ. Safety.* 7,400-409.

Hose, J.E., J.B. Hannah, M.L. Landolt, B.S. Miller, S.P. Felton, and W.T. Iwaoka. 1981. Uptake of benzo(a)pyrene by gonadal tissue of flatfish (family Pleuronectidae) and its effects on subsequent egg development. *J. Toxicol. Environ. Health.* 7:991-1000.

Hose, J.E., J.B. Hannah, D. Dijulio, M.L. Landolt, B.S. Miller, W.T. Iwaoka, and S.P. Felton. 1982. Effects of benzo(a)pyrene on early development of flatfish. *Arch. Environ. Contam. Toxicol.* 11:167-171.

Hrudey, S.E., W. Chen, and C.G. Rousseaux. 1996. Bioavailability in environmental risk assessment. CRC Press, Inc., Boca Raton, Florida, pp.294.

Jarman, W; K. Hobson, W. Sydeman, C. Bacon and E. McLaren. 1996. Influence of Trophic Position and Feeding Location on Contaminant Levels in the Gulf of the Farallones Food Web Revealed by Stable Isotope Analysis. *Environmental Science & Tech.* 30(2):654-660.

Landrum P.E., B.J. Eadie, and W.R. Faust. 1988. Toxicity and toxicokinetics for a mixture of sediment associated polycyclic aromatic hydrocarbons to the amphipod Pontoporeia hoyi. *In:* Poster Abstracts, SETAC Ninth Annual Meeting, Society of Environmental Toxicology and Chemistry, Washington D.C. p. 29.

Lee, R.F., J. Stolzenbach, S. Singer, and K.R. Tenore. 1981. Effects of crude oil on growth and mixed function oxygenase activity in polychaetes, Nereis sp. *In:* Biological Monitoring of Marine Pollutants. Ed. Vernburg, F.A. Calabrese, F. Thurberg, and W. Vernberg. Academic Press. pp. 323-334.

Lee, H., II, Boese, B.L., Pelletier, J., Winsor, M., Specht, D.T., and Randall, R.C., 1989. Guidance Manual: Bedded Sediment Bioaccumulation Test. USEPA Pacific Ecosystem Branch Bioaccumulation Team, Newport, OR.

Lee, H., II, Lincroft, A, et al, 1994. Ecological Risk Assessment of the Marine Sediments at the United Heckathorn Superfund Site. USEPA Pacific Ecosystem Branch Bioaccumulation Team, Newport, OR., USEPA Region IX, San Francisco, CA.

Lunde, G. 1977. Occurrence and transformation of arsenic in the marine environment. *Environmental Health Perspectives* 19: 47-52.

McCarty, L.S. 1986. The relationship between aquatic toxicity QSARs and bioconcentration for some organic chemicals. *Environ. Toxicol. Chem.* 8:1071-1080.

McCarty, L.S. 1991. Toxicant body residues: implications for aquatic bioassays with some organic chemicals. *In:* Aquatic Toxicology and Risk Assessment: Fourteenth Volume, ASTM STP 1124; M.A. Mayes and M.G. Barron, Eds., American Society for Testing and Materials, Philadelphia; pp. 183-192.

McCarty, L.S., D. MacKay, A.D. Smith, G.W. Ozburn, and D.G. Dixon. 1992. Residue-based interpretation of toxicology bioconcentration QSARs from aquatic bioassays: neutral narcotic organics. *Environ. Tox. Chem.* 11: 917-930.

McElroy A.E., J.M. Cahill, J.D. Sisson, and K.M. Kleinow. 1991. Relative bioavailability and DNA adduct formation of Benzo[a]pyrene and metabolites in the diet of the winter flounder. *J. Comp. Biochem. Physiol.* 100, 12-29.

McElroy. A.E. and J.D. Sisson. 1989. Trophic transfer of Benzo[a]pyrene metabolites between benthic marine organisms. *Mar. Environ. Res.* 28, 265-269.

McElroy, A.E., J.M. Cahill, J.D. Sisson, and K.M.Kleinow. 1991. Relative bioavailability and DNA adduct formation of Benzo[a]pyrene and metabolites in the diet of the winter flounder. *J. Comp. Biochem. Physiol.* 100C:1-2,29-33.

McFarland, V.A. 1995. Evaluation of Field-Generated Accumulation Factors for Predicting the Bioaccumulation Potential of Sediment-Associated PAH Compounds. USACE - WES Technical Report D-95-2. July 1995.

Meador J.P., J.E. Stein, W.L. Reichert, and U.Varanasi. 1995. Bioaccumulation of polycyclic aromatic hydrocarbons by marine organisms. *Rev. Environ. Contam. Toxicol.* 143, 79-165.

Naqvi, S.M., Flagge, C.T., and Hawkins, R.L. 1990. Arsenic uptake and depuration by Red Crayfish, *Procambarus clarkii*, exposed to various concentrations of monosodium methanearsonate (MSMA) herbicide. *Bull. Environ. Contam. Toxicol.*, 45:94-100.

O'Connor, J.M., A.R.Schnitz, and K.A. Squibb. 1988. In vivo kinetics of Benzo[a]pyrene and 7,12-dimethylbanz[a]anthracene assimilation and metabolism in rainbow trout. *Mar. Environ. Res.* 24:63-67.

Oladimeji, A.A., Qadri, S.U., and deFreitas, S.W. 1984. Long-term effects of arsenic in rainbow trout, *Salmo gairdneri*. *Bull. Environ. Contam. Toxicol.*, 32:732-741.

Parrish, P.R., S.C.Schimmel, D.J. Hansen, J.M. Patrick, J. Forester. 1976. Chlordane: Effects on Several Estuarine Organisms. *Journal of Toxicology and Environmental Health*, 1:485-494.

Pruell, R.J., N.I. Rubinstein, B.K. Taplin, J.A. LiVolsi, R.D. Bowen. 1993. Accumulation of polychlorinated organic contaminants from sediment by three benthic marine species. *Arch. Envir. Contam. Toxicol.* 24, 290-297.

Rice, D.R., M.M. Babcock, C.C. Brodersen, J.A. Gharrett and S. Korn. 1987. Uptake and depuration of aromatic hydrocarbons by reproductively ripe pacific herring and the subsequent effect of residues on egg hatching and survival. *In:* Pollution Physiology of Estuarine Organisms. Ed. Vernberg, W., A. Calabrese, F. Thruberg, and F. Vernberg. University of South Carolina Press. pp. 139-154.

Riedel, G.F., Sanders, J.G., and Osman, R.W. 1987. The effect of biological and physical disturbances on the transport of arsenic from contaminated estuarine sediments. *Estuarine, Coastal and Shelf Science*, 25:693-706.

Rubinstein, N.I., Lores, E., and Gregory, N.R. 1983. Accumulation of PCBs, mercury and cadmium by *Nereis virens, Mercenaria mercenaria* and *Palemonetes pugio* from contaminated harbor sediments. *Aquatic Toxicol.*, 3:249-260.

Rubinstein, N. I., R. J. Pruell, B. K. Taplin, J. A. LiVolsi, and C. B. Norwood. 1990. Bioavailability of 2,3,7,8-TCDD, 2,3,7,8-TCDF, and PCBs to marine benthos from Passaic River sediments. *Chemosphere*, 20, 1097-1102.

Squibb, K.S., J.M. O'Connor, and Kneip, T.J. 1991. Toxics Characterization Report, Module 3.1. Report prepared by Institute of Environmental Medicine, NY Univ. Medical Center for the NY/NJ Harbor Estuary Program.

Steimle, F.W., V.S. Zdanowicz, S.L. Cuneff and R. Terranova. 1994. Trace metal concentrations in common benthic macrofaunal prey form the New York Bight. US National Marine Fisheries Service, NOAA. *Marine Pollution Bulletin.* 28, 12, pp. 760-765.

Suedel, B.C., J.A. Boraczek, R.K. Peddicord, P.A. Clifford, and T.M. Dillon. 1994. Trophic transfer and biomagnification potential of contaminants in aquatic ecosystems. *Reviews of Environmental Contamination and Toxicology* 136: 21-89.

Sweeney, B., D. Funk and L. Standley. 1993. Use of the Stream Mayfly Cloeon Triangulifer as a Bioassay Organism: Life History Response and Body Burden Following Exposure to Technical Chlordane. *Environ. Tox. and Chem.* 12:115-125.

Syracuse Research Corporation, Environmental Science Center. 1996. Experimental Log P (Octanol/water partition coefficient database). http://esc.syrres.com/~ESC/kowexpdb.htm

Thomas, L.M. 1987. Letter from Lee M. Thomas, Administrator, U.S. Environmental Protection Agency to Honorable Henry A. Waxman, Chairman, Subcommittee on Heath and the Environment, Committee on Energy and Commerce, House of Representatives. May 29, 1987.

USACE. 1981. Final Interpretive Guidance for Bioaccumulation of Petroleum Hydrocarbon, DDT, Cadmium, and Mercury in the New York Bight. Memorandum from North Atlantic Division Corps of Engineers to G.R. Tobertson, Deputy Director of Civil Works, Dept. of Army.

USACE. 1994. Bioaccumulation Guidance Values for Selected Contaminants in Sediments and Biota of the Sandy Hook Reference Site for the New York Bight Apex Mud Dump Site. (draft) Report by Corps of Engineers Waterways Experiment Station (WES) for the New York District Corps.

USACE. 1995. Trophic transfer and biomagnification potential of contaminants in aquatic ecosystems. *In:* Environmental Effects of Dredging Technical Notes. EEDP-01-33. USACE Waterways Experiment Station (WES).

USEPA/CENAN. 1992. Guidance for Performing Tests on Dredged Material Proposed for Ocean Disposal. New York District Corps of Engineers, U.S. Environmental Protection Agency -Region 2.

USEPA/CENAN. 1997. (Joint Memorandum) Ocean Disposal of Dredged Material Clarification of Two Procedural Elements of Interagency Coordination Between USEPA Region 2 and the New York District, USACE-Treatment of Non-Detects, Chemical Data, and Rules and Responsibilities in Preparation of Ocean Disposal Regulatory Compliance Memorandum.

USEPA/USACE. 1991. Evaluation of Dredged Material Proposed for Ocean Disposal - Testing Manual. (Green Book). EPA - 503/8-91/001.

USEPA. 1980a. Water quality criteria documents: availability. *Federal Register*, Vol. 45, No. 231. November 28, 1980.

USEPA. 1980b. Ambient Water Quality Criteria for Aldrin/Dieldrin; EPA 440/5-80-019; December 1980.

USEPA. 1980c. Ambient Water Quality Criteria for Chlordane; EPA 440/5-80-027; October 1980.

USEPA. 1980d. Ambient Water Quality Criteria for Heptachlor; EPA 440/5-80-052; October 1980.

USEPA. 1980e. Ambient Water Quality Criteria for Endosulfan; EPA 440/5-80-046; October 1980.

USEPA. 1980f. Ambient Water Quality Criteria for Dichlorobenzenes; EPA 440/5-80-039; October 1980.

USEPA. 1984a. Ambient Water Quality Criteria for Lead - 1984; EPA 440/5-84-027; January 1985.

USEPA. 1984b. Ambient Water Quality Criteria for Copper - 1984; EPA 440/5-84-031; January 1985.

USEPA. 1985a. Ambient Water Quality Criteria for Chromium - 1984; EPA 440/5-84-029; January 1985.

USEPA. 1985b. Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses. NTIS # PB85-227049.

USEPA. 1985c. Ambient Water Quality Criteria for Arsenic - 1984; EPA 440/5-84-033; January 1985.

USEPA. 1986. Ambient Water Quality Criteria for Nickel - 1986; EPA 440/5-86-004; September 1986.

USEPA. 1987a. National primary drinking water regulations - synthetic organic chemicals; monitoring for unregulated contaminants; final rule. *Federal Register*, Vol. 52, No. 130, 25690. July 8, 1987.

USEPA. 1987b. Ambient Water Quality Criteria for Zinc - 1987; EPA 440/5-87-003.

USEPA. 1988. Guidance for state implementation of water quality standards for CWA section 303(c)(2)(B). *Federal Register*, Vol. 54, No. 346. November 12, 1988.

USEPA. 1989. Interim Procedures for Estimating Risks Associated with Mixtures of Chlorinated Dibenzo-p-Dioxins and -Dibenzofurans (CDDs and CDFs) and 1989 Update. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC. EPA/625/3-89/016.

USEPA. 1991. National Primary Drinking Water Regulations; Final Rule. 40 CFR Part 141. January 30, 1991.

USEPA. 1992a. Draft Ambient Water Quality Criteria for Silver.

USEPA. 1992b. Water quality standards; establishment of numeric criteria for priority toxic pollutants; states compliance. *Federal Register, Vol.* 57: 60848.

USEPA. 1993. Provisional Guidance for Qualitative Risk Assessment of Polycyclic Aromatic Hydrocarbons. EPA/600/R-93/089.

USEPA. 1994. Final clarification of suspended particulate phase bioaccumulation testing requirements for material dumped in ocean waters. *Federal Register* Vol. 59: 52650. October 18, 1994.
USEPA. 1995. On-Line. Integrated Risk Information System (IRIS). Cincinnati, OH: Office of Research and Development, Environmental Criteria and Assessment Office.

USEPA. 1996a. Ocean dumping testing requirements; final rule. *Federal Register*, Vol. 61, No. 190, 51196. September 30, 1996.

USEPA. 1996c. Memo to File from A. Lechich. Subject: Issues Regarding Exposure and Uptake Mechanisms for PAHs. (Discussion with V. McFarland). December 5, 1996.

USEPA. 1996d. Memo to File from A. Lechich. Subject: Discussion of PAHs With Regard to East River Memo. (Discussion with D. Hansen). December 5, 1996.

USEPA. 1996e. Memo to File from C. Vogt. Subject: Acceptable Levels of Lead: East River Bioaccumulation Tests. December 13, 1996.

USEPA. 1996f. Battelle Body Burden Study. Report prepared by Battelle Ocean Sciences, Duxbury, MA, for USEPA - Region II.

USEPA. 1997a. Memo to File from A. Lechich. Subject: Summary of Dioxin Risk Evaluation Approach. March 15, 1997.

USEPA. 1997b. Contaminants in Polychaetes from the Mud Dump Site and Environs. March 4, 1997. Report prepared by Battelle Ocean Sciences, Duxbury, MA, for USEPA - Region II.

USEPA. 1997c. Supplemental to the Environmental Impact Statement on the New York Dredged Material Disposal Site Designation for the Designation of the Historic Area Remediation Site (HARS) in the New York Bight Apex. U.S. Environmental Protection, Region 2, New York, May 1997.

Varanasi U., J.E. Stein, and M. Nishimoto. 1989. Biotransformation and disposition of polycyclic aromatic hydrocarbons (PAH) in fish. *In:* Varanasi U. (ed) Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment. CRC Press, Boca Raton, Fl, pp 94-149.

V-Balogh, K., and Salanka, J. 1984. The dynamics of mercury and cadmium uptake into different organs of *Anodonta cygnea* L. *Water res.*, 18(11):1381-1387.

Veith, G. D., DeFoe, D.L., and Bergstedt, B.V. 1979. Measuring and estimating the bioconcentration factor of chemicals in fish. *J. Fish. Res. Board Can.*, 36(9):1040-1048.

Verschueren, K. 1983. Handbook of Environmental Data on Organic Chemicals, second edition. Van Nostrand Reinhold Company.

Ward, G.S., P.R. Parrish, and R.A. Rigby. 1981. Early life stage toxicity tests with a saltwater fish: Effects of eight chemicals on survival, growth, and development of sheepshead minnows (Cyprinidon variegatus). *J. Toxicol. Environ. Health.* 8:225-240.

Whittle K.J., J. Murray, P.R. Mackie, R. Hardy, and J. Farmer. 1977. Fate of hydrocarbons in fish. *In:* Petroleum Hydrocarbons in the Marine Environment. *Cons. Intern. Explor. Mer.* Vol. 171, McIntyre A.D. and Whittle K.J. (eds), pp 139-142. Charlottenlund Slot, Denmark.

WHO. 1993. Guidelines for Drinking Water Quality. World Health Organization. Geneva.

TABLE 1

Bioaccumulation Table for Region 2 Projects, ALL VALUES ARE IN WET WEIGHT 0.044

Project Name: 2011 Arecibo Harbor Federal Navigation Project

Template Version: 12/27/00

	PROJECT DATA												COMPARISON DATA						
Col. 1	Col. 2 [11]	Col. 3 [11]	Col. 4[11]	5	Col. 6[11]	7	Col. 8	Col. 9	Col. 10	Col. 11	Col. 12	Col. 13	Col. 14	Col. 15	Col. 16	Col. 17	Col. 18	Col. 19	Col. 20
Sample I.D.	Reference	Reference	Test Sed.	[1]	Test Sed.	[1]	Conv.Fac.	Test Sed.	Test Sed.		Test Sed.	Test Sed.	Human Health	Human Health				Ecological	
							clam/worm	SS	SS	Carcinogenic	BaP Tox. Equiv.	BaP Tox. Equiv.	Cancer (10E-4)	Non-Cancer	Background	Background	FDA	Non specific	Regional
	(clam)	(worm)	(clam)		(worm)		[2]	(clam)	(worm)	TEF [3]	Conc.(clam)[4]	Conc.(worm)[4]	Level[5]	Level (HQ=1)	(clam)	(worm)	Limits[7]	Effects Level	Matrix[12]
Compound	(ug/Kg)	(ug/Kg)	(ug/Kg)		(ug/Kg)			(ug/Kg)	(ug/Kg)		(ug/Kg)	(ug/Kg)	(ug/Kg)	(ug/Kg)[10]	(ug/Kg)	(ug/Kg)	(ug/Kg)	(ug/Kg)	(ug/Kg)
PAHs																			
Acenaphthene			0.222		0.5		1/1	0.22	0.50					8,775,000	8.1	0.5		[15]	
Acenapthylene			0.06		0.31		1/1	0.06	0.31					[6]	4.6	1.3		[15]	
Anthracene			0.16		0.15		1/1	0.16	0.15					43,605,000	10.0	1.6		3,750	
Fluorene			0.33		0.61		1/1	0.33	0.61					5,805,000	7.4	0.3		[15]	
Naphthalene			1.02		2.13		1/1	1.02	2.13					[6]	26.4	4.5		[15]	
Phenanthrene			0.94		1.11		1/1	0.94	1.11					43,605,000	32.7	4.7		[15]	
Benzo(a)anthracene*			0.2		0.044		2/2	0.40	0.09	0.1	0.04	0.0088		[6]	25.1	4.8		[15]	
Benzo(a)pyrene*			0.15		.015U		2/2	0.30	0.03	1	0.3	0.03		[6]	24.2	7.6		8000 [8]	
Benzo(g,h,i)perylene			0.12		0.065		3/3	0.36	0.20					[6]	22.2	7.6		[15]	
Benzo(b)fluoranthene*			0.35		.01U		2/2	0.70	0.02	0.1	0.07	0.002		[6]	40.6	16.5		[15]	
Benzo(k)fluoranthene*			0.33		.015U		2/2	0.66	0.03	0.01	0.0066	0.0003		[6]	81.6	5.6		[15]	
Chrysene*			0.61		0.52		2/2	1.22	1.04	0.001	0.00122	0.00104		[6]	29.4	6.7		[15]	
Dibenzo(a,h)anthracene*			.02U		.02U		2/2	0.04	0.04	1	0.04	0.04		[6]	8.5	1.1		[15]	
Fluoranthene			1.67	X	0.81		1/1	1.67	0.81					5,805,000	43.8	10.6		[15]	
Indeno(1,2,3-cd)pyrene*			0.063		.035U		3/3	0.19	0.11	0.1	0.0189	0.0105		[6]	16.6	4.4		[15]	
[illegible]			2.53	X	0.66		1/1	2.53	0.66					4,387,000	51.3	26.6		[15]	
TOTAL PAHs								10.80	7.83		0.47672	0.09264	2,000(BaP.eqv.)		432.7	104.60		40000 [9]	
PESTICIDES																			
Aldrin			.03U		.03U		2/2	0.06	0.06				33	167	0.9	0.1	300	299[10]	
Dieldrin			0.06	X	0.13		2/2	0.12	0.26				65	518		0.1	300	4.37[10]	
a-Chlordane			0.09	X	0.09		2/2	0.18	0.18							0.7	300		
Trans nonachlor			.03U		0.21	X	2/2	0.06	0.42							0.5			
Heptachlor			.04U		.04U		2/2	0.08	0.08							0.05	300		
Heptachlor epoxide			.035U		.035U		1/1	0.04	0.04							0.2	300		
Total Residual Chlordane/Heptachlor								0.36	0.72				114	135		1.7		64#[10]	
Endosulfan I			.035U		.035U		1/1	0.04	0.04				[6]			0.2			
Endosulfan II			.025U		.025U		1/1	0.03	0.03				[6]			0.1			
Endosulfan sulfate			.06U		.06U		1/1	0.06	0.06				[6]			0.1			
Total Endosulfans								0.12	0.12					87,000		0.4		2.86#[10]	
4,4-DDT			.03U		0.06		11/11	0.33	0.66						0.6	1.0			
2,4-DDT			.015U		.015U		2/2	0.03	0.03							0.4			
4,4-DDD			.04U		0.18		3/3	0.12	0.54							3.9			
2,4-DDD			0.05	X	0.178		2/2	0.10	0.36							1.4			
4,4-DDE			0.14	X	0.06		2/2	0.28	0.12						3.5	4.3			
2,4-DDE			.03U		.03U		2/2	0.06	0.06							0.1			
Total DDT								0.92	1.77							11.1			40
TOTAL PCBs			9.75	X	24.61	X	1/2	9.75	49.22						106.6	88.1	2,000		100(clam)113(worm)[18]
METALS								(mg/Kg)	(mg/Kg)	(mg/Kg)	(mg/Kg)	(mg/Kg)	(mg/Kg)	(mg/Kg)	(mg/Kg)	(mg/Kg)	(mg/Kg)	(mg/Kg)	(mg/Kg)
Arsenic			4.51		2.04		1/1	4.51	2.04				[17]	[17]		4.89		12.6[10]	
Cadmium			0.05		0.027	X	[16]	0.05	0.027						1.21	0.11			0.3
Chromium (total)			1.2	X	0.077		1/1	1.2	0.077				[6]	73	1.28	1.31		11.8[10]	
Copper			2.96	X	16.2	X	1/1	2.96	16.2				[6]	540	5.58	2.78		9.6[10]	
Lead			0.318	X	0.147	X	1/1	0.318	0.147				[6]	1.3	1.41	1.64		11.9[10]	
Mercury			0.012	X	0.026		[16]	0.012	0.026						0.04	0.03	1		0.2
Nickel			0.82	X	0.08		1/1	0.82	0.08				[6]	290	1.10	0.77		3.8[10]	
Silver			0.037		0.017	X	1/1	0.037	0.017				[6]	73		0.15		1.4[10]	
Zinc			18		14.4		1/1	18	14.4				[6]	4,400	11.50	20.61		1517[10]	

Appendix for Table 1

I. CONSIDERATION OF ECOLOGICAL EFFECTS

A. Potential for ecological effects based on Water Quality Criteria (Column 19)

The potential for ecological impacts due to bioaccumulation of several compounds of concern was evaluated by calculating a Water Quality Criterion Tissue Level (WQCTL). The WQCTL is calculated by multiplying the Clean Water Act Section 304(a)(1) Federal water quality criterion chronic value (CV) for the chemical by the empirically determined bioconcentration factor (BCF) for the chemical for a representative marine organism (Lee, *et al.*, 1989). A BCF is the ratio of the concentration of a contaminant in an organism to the concentration of the contaminant in water. Thus, the WQCTL represents the tissue concentration that would be expected in an organism exposed to water containing the chemical at the CV concentration. This level is set to protect 95% of all tested organisms included in the water quality criterion database, thus representing a conservative level of protection (USEPA, 1985b). Table 1 lists the calculated WQCTLs. Sources of CVs and BCFs are from USEPA ambient water quality criteria documents (USEPA 1980b, 1980c, 1980d, 1980e, 1980f, 1984a, 1984b, 1985a, 1985c, 1986, 1987b and 1992a) and Calabrese (1984)(for silver). Calculations are shown in attachment A.

Several pesticides were evaluated based on the sum of their primary constituents and associated metabolites (e.g., total chlordane, total endosulfan, and total DDT). Alpha(trans)-chlordane, trans nonachlor, heptachlor and heptachlor epoxide represent the primary components of technical chlordane and its metabolites found in the tissue of aquatic organisms (Jarman, et. al., 1996; Verschueren, 1983; Sweeney, et. al., 1993). These constituents are summed as total chlordane as is consistent with current practice for chlordane (Jarman, et. al., 1996) and total DDT. The WQCTL for total chlordane was calculated using the WQC for chlordane as a conservative level of protection. While water quality criteria exist, and WQCTLs can be calculated, for heptachlor (133 ppb) and chlordane (64 ppb), the sum total chlordane is compared to the WQCTL for chlordane in order to be more environmentally conservative. The chlordane WQCTL provides a conservative level of protection as indicated by published residue effects levels (Sweeney, et. al., 1993; Bauman, et. al., 1987; Feroz, et. al., 1979; Parrish, et. al., 1976). Consistent with the above approach, the tissue concentration for endosulfan I, endosulfan II and endosulfan sulfate were also summed as total endosulfan and compared to the WQCTL for total endosulfan.

The WQCTLs were also calculated for all metals of concern which don't have Matrix values. For total chromium, the WQCTL was calculated based on chromium(VI), which is substantially more toxic than chromium (III) and elemental chromium in order to provide a conservative level of environmental protection.

B. Potential for ecological effects based on PAH toxicity (Column 19).

The Critical Body Residue (CBR) approach described by McCarty (1991) was used to derive values for use in evaluating the potential impacts of PAHs accumulated in the dredged material bioaccumulation test organisms. CBRs are concentrations of chemical residues in organisms which elicit a deleterious biological response associated with narcosis, which is the primary non-cancer effect of PAHs. Narcotic responses measured can be acute (e.g., immobilization or death) or chronic endpoints (e.g., reduced reproduction, fecundity or growth). CBRs are represented as the ratio of the mass of toxicant to the mass of the organism, such as millimoles or micrograms of toxicant per kilogram (mmole or ug/kg) of organism. For the narcosis endpoint, each molecule of individual PAH congeners are generally equipotent, thus the total PAH concentration is compared to the CBR. For example, a 400 ppb dose of naphthalene would elicit a similar toxicity response as 400 ppb of fluorene; if both chemicals are present together at these concentrations, then the dose would equal 800 ppb.

McCarty (1991) states that an average critical body residue of 400,000 - 1,200,000 ppb can be used as an estimate for acute effects for a narcosis-producing chemical (e.g., PAHs) on fish populations. (Note: McCarty reports the CBR in units of millimoles per kilogram; this value has been converted to ppb for PAHs using the average molecular weight of the PAHs analyzed in the bioaccumulation test). Chronic effect critical body residues can be estimated by applying an acute to chronic ratio of 10 to the acute CBR (McCarty, 1986; Call, *et al.*, 1985). Therefore, the chronic critical body residue for PAHs can be estimated at 40,000 - 120,000 ppb of PAHs in organism tissue, and Table 1 thus uses the 40,000 ppb level.

These CBRs were based on fish data. The use of CBRs based on fish toxicity represents a conservative estimate of potential toxicity due to exposure to dredged material because: (1) it is extremely unlikely that a fish would get its whole diet from the HARS; and (2) fish are generally more sensitive than the benthic organisms in direct contact with the dredged material placed at the HARS (e.g., Landrum, *et al.* (1988) estimated an acute CBR for crustaceans of 800,000 ppb - 42,000,000 ppb).

C. Potential ecological impacts of mutagenic, carcinogenic and teratogenic PAHs (Column 19)

USEPA and the USACE reviewed eleven scientific journal articles to obtain information about the potential for adverse effects to the marine environment due to the observed bioaccumulation of PAHs in the marine worm, *Nereis virens*, and the clam, *Macoma nasuta*. These articles reported the results of laboratory experiments that sought to relate the concentration of a contaminant(s) in water, as injected doses, or tissue concentrations, to mutagenic, carcinogenic, teratogenic and/or reproductive effects to fish. These studies all used fish species which are considered to be among the most sensitive organisms in the marine environment to exhibit the above effects (USEPA, 1996c). In addition, most of these studies focussed on the PAH most believed to cause such effects for which there is data, benzo(a)pyrene (BaP). One study (Breteler, 1984), discussed the possible sources and distribution of PAHs in the Hudson/Raritan estuary, and ranked the threat of PAHs to aquatic biota and humans. The main threat was

believed to be carcinogenicity, with a greater threat ranking assigned to humans than biota. However, Breteler (1984) did not provide specific effects-based levels that could be used in the following analysis. Two articles evaluated the effects of crude oil, and thus were not useful for evaluating the effects of specific PAHs measured in the bioaccumulation test (Rice, *et al.*, 1987; Lee, *et al.*, 1981). Three studies considered the effects of specific PAHs, but did not synoptically measure tissue concentrations in the organisms (Ward, *et al.*, 1981; Holcombe, *et al.*, 1983; Finger, *et al.*, 1985) and were not used, because the lack of tissue data for these studies makes their utility in evaluating the tissue concentration resulting from the dredged material bioaccumulation tests highly uncertain.

The remaining five papers reported measured tissue concentrations and observed reproductive effects in organisms exposed to PAH-spiked water. One article reported the tissue concentrations of adult fish and the observed effect on survival and health of the fish's offspring. Hose, *et al.* (1981) reported that adult English sole injected with benzo(a)pyrene (BaP) accumulated the chemical in the gonad and mature gametes. The amount of BaP taken up by the ovary ranged from 16,800 to 49,700 ppb. Two samples of ripe eggs contained 51,200 and 263,000 ppb of BaP and its metabolites. No adverse effects were reported for these concentrations. Hose, *et al.* (1981) also reported the results of injecting female flathead sole with BaP. Adverse effects to egg hatching success were reported for each female. The paper does not report tissue concentrations in either the parent fish or the egg of the flathead sole. Effects on reproductive success were reported but could only be correlated with the external dose injected into the parent. Therefore, since concentrations and effects were not synoptic in this report, it was not useful in the evaluation of the dredged material bioaccumulation results.

Three papers reported the results of experiments which measured fish egg or alevin concentrations of BaP and associated reproductive or carcinogenic effects (Hose, *et al.*, 1982; Hannah, *et al.*, 1982; Black, *et al.*, 1988). Hose, *et al.* (1982) exposed three species of sole, sand sole, English sole, and flathead sole, to BaP-spiked water. Tissue concentrations of 2,100 ppb were measured in sand sole on day 6 (24 hours after hatching) and were associated with reduced hatching success. However, we did not consider the results to be appropriate for use in setting effects levels because they may have been compromised by the methods of replication used in the experimental design.

Hannah, *et al.* (1982) estimated a concentration of BaP in tissue that caused abnormalities in development of rainbow trout eggs, using aqueous exposures and actual measured tissue concentrations in alevin tissues. An exposure to a 2.4 ppb mean aqueous BaP concentration accumulated an average of 12,340 ppb in alevins. This concentration was associated with an increase in percentage of abnormalities from approximately 6% at lower water concentrations (0.08, 0.21, 0.37 and 1.48 ppb) to approximately 13% at higher concentrations. From 0.08 to 1.48 ppb in the water, there were no increasing effects exhibited, therefore, the effects were apparently "real" (i.e., significantly greater than the threshold effect level of 6%) only at the aqueous 2.4 ppb concentration. The Hannah, *et al.*,(1982) study is considered the most reliable study for this evaluation since it used exposure series and measured tissue concentrations associated with observed effects, and therefore allows for the calculation of a no-effects level

directly from the measured results.

In applying these studies to evaluations of dredged material, consideration must be given to uncertainties in converting these kinds of results to concentrations protective of other biota. Three uncertainties needing to be considered are: (1) those associated with converting effect to no-effect concentrations, (2) across-species uncertainties, and (3) uncertainties in estimating the dose of contaminants to which the organism is exposed. These uncertainties are discussed below.

With respect to uncertainty when converting effect to no-effect concentrations, an uncertainty factor of one order of magnitude is often used when only an effect measure is reported. However, in Hannah, *et al.*(1982), the no-effect level can be estimated to be the next lowest concentration below the lowest-observed effect level, since the range of concentrations below this level did not exhibit significantly different responses. In this case, the no-effect level occurred at the water exposure concentration of 1.48 ppb. Although a tissue concentration was not measured at the 1.48 ppb water concentration, it can be calculated from the concentration measured at the effect level (i.e., the no-effect water concentration (1.48 ppb) is close to 65 percent of the observed effect concentration (2.4 ppb) so the no-effect tissue concentration should be about 65 percent of the lowest-observable effect tissue concentration (0.65 x 12,340 ppb = 8,021 ppb)). Thus, a factor to adjust these data from lowest observed effect tissue concentration to the calculated no-observed effect tissue concentration is obtained directly from the data.

There can also be uncertainty as to the proximity to the site of toxic action in the organism that a dose or concentration is measured, and with respect to species-to-species variability. Hannah (1982) reported dose concentrations in the tissue and, therefore, there is no need to account for variability associated with the large uncertainties encountered in typical water-only exposure studies where the actual concentration at the site of toxic action is unknown. When measured in the tissues, as was done for this project, concentrations of narcotic chemicals causing effects (i.e., critical body residues, CBRs) in aquatic organisms are reported to range only from 1.4 to 21 umoles/g wet weight (a factor of about one order of magnitude) for organisms as diverse as insects, crustaceans, and fish (McCarty, *et al.* 1992). Therefore, from a tissue concentration perspective, the species-to-species uncertainty factor appropriate for both total PAHs operating as narcotics and individual PAHs having teratogenic effects would be one order of magnitude, or a value of 10 (USEPA, 1996d).

In summary, a factor of 10 (representing species to species uncertainty) is an appropriate UF to use in these evaluations. Also, as described in memo SectionV, subsection C2(c)(i) above, Brown and Neff (1993) show that trophic transfer of PAHs up the food chain to fish decrease tissue levels by over an order of magnitude. Given this data and the fact that these studies included fish that spent 100% of their time feeding in the test sediment, whereas this would be highly unlikely to occur at an ocean site, a trophic transfer factor of 0.1 is used in this analysis. Applying this UF of 10 and a trophic transfer factor of 0.1 to the no-effects level for BaP calculated from Hannah, *et al.* (1982), as discussed above (8,021 ppb) results in a no-effect level

for BaP of approximately 8,000 ppb in benthic tissue.

II. CONSIDERATION OF POTENTIAL EFFECTS ON HUMAN HEALTH (Columns 14 and 15)

Human effects screening levels were developed with risk-based methods using conservative estimates of exposure to assess whether these contaminants would accumulate to levels in fish and shellfish that could lead to significant adverse effects to humans. The approach assessed consumption of fish and shellfish to derive conservative estimates of contaminant concentrations in benthic tissue protective of human health using the following USEPA standard risk-assessment assumptions: a 70-kilogram adult eats 6.5 grams of fish and shellfish per day over a 70-year lifetime. This assessment considered potential for both cancer and non-cancer effects in humans. USEPA IRIS (USEPA, 1995) and effects information from USEPA's National Toxics Rule (USEPA, 1992b) were used in the human health assessment to calculate acceptable levels in fish and shellfish to protect human health. Trophic transfer factors, as discussed earlier, were then used to convert these fish and shellfish levels into benthic tissue concentrations.

For regulatory purposes, USEPA utilizes 10^{-4} to 10^{-6} (one in ten thousand to one in one million) as an acceptable incremental risk range for activities with potential for causing cancer in human beings (USEPA, 1980a; USEPA, 1988; USEPA, 1987a; Thomas, 1987; USEPA, 1991). USEPA considers a cancer risk within this range to be safe and protective of public health. This is supported by the World Health Organization Guidelines for Drinking Water Quality (WHO, 1993), where it selected a 10^{-5} guideline value, and then explained that the application could vary by a factor of ten (e.g., 10^{-4} to 10^{-6}). Since this analysis uses conservative methods, the results represent conservative estimates of risk, or what are in effect plausible upper-bound estimates. Thus, the true risk is highly unlikely to be greater than estimated and could be much lower.

Table 1, Column 14 lists human cancer protection levels in benthic organisms for chemicals which are known or suspected carcinogens that would lead to a human cancer risk level of 10^{-4}. For PAHs, this analysis used BaP-equivalents derived from the toxic equivalence factor for each carcinogenic PAH (from USEPA (1993); note: these factors are listed in Column 11 of Table 1 for each of the compounds).

The potential for non-cancer impacts can be expressed as a hazard quotient (HQ), which is the ratio of the average daily intake divided by the toxicological reference dose for the chemical. If the HQ is less than unity (e.g., 1), an adverse noncarcinogenic effect is highly unlikely to occur. If the HQ exceeds unity, an adverse health impact may occur. The higher the HQ, the more likely that an adverse noncarcinogenic effect will occur as a result of exposure to the contaminant in the dredged material after placement. Table 1, Column 15 lists noncancer protection levels in benthic organisms for the chemicals that are known to cause, or suspected of causing, non-carcinogenic effects, that would result in a human HQ equal to unity. Those numbers were derived using the conservative assumptions and source materials described in the introductory paragraph to this section.

For the following compounds, the following special considerations were used in evaluating the results in Table 1.

Metals

No reference dose has been established for lead. EPA has adopted a blood lead level of 10ug/dl as the level of concern and EPA policies are that regulatory programs should seek to minimize the number of children with blood lead levels above a target of 10 ug/dl (Final Rule for Lead and Copper NPDWR, 56FR26468, June 7, 1991), and this value was used to calculate the effects level in Table 1 (see USEPA 1996e).

When interpreting the importance of arsenic tissue concentrations for human health, consideration was given to the arsenic form present (i.e., inorganic vs. organic). Arsenic is found in marine organisms as an organic complex which includes such compounds as arsenobetaine and arsenocholine (Abel and Axiak, 1991). Organic arsenic in the tissues of aquatic organisms is not metabolized by predators or humans and is readily eliminated from the body through excretion (Hrudey *et al.*, 1995). As a result, the toxicity of organic arsenic ingested from seafood is low and appears to pose no significant hazard (Abernathy and Ohanian, 1992). For this reason, cancer and non-cancer protection levels, based on inorganic arsenic as contained in EPA's IRIS database, are not appropriate for evaluating the potential human health impacts of arsenic bioaccumulation from dredged material, and therefore, are not included in Table 1.

Pesticides

Alpha(trans)-chlordane, trans nonachlor, heptachlor and heptachlor epoxide represent the primary components of technical chlordane and its metabolites found in the tissue of aquatic organisms (Jarman, et. al., 1996; Verschueren, 1983; Sweeney, et. al., 1993). These constituents are summed as total chlordane as is consistent with current practice for chlordane (Jarman, et. al., 1996) and other pesticides (e.g., total DDT). Total chlordane is evaluated using the 10^{-4} cancer risk level and non-cancer level for heptachlor epoxide, which has the greatest potency of the chlordane constituents or metabolites. Similarly, endosulfan I, endosulfan II, and endosulfan sulfate are summed and the total is compared to the conservative non-cancer protection level for endosulfan.

PAHs

For PAHs, this analysis used BaP-equivalents derived from the toxic equivalence factor for each carcinogenic PAH (from USEPA (1993); note: these factors are listed in column 11 of Table 1 for each of the compounds).

Attachment A:
Tissue concentration is calculated using BCF * Water Quality Criteria (WQC) ambient aqueous concentration, and assuming the conversion factor of 1 kg=1L.

Compound	**Ambient Conc. (ug/L)[1]**	**BCF**	**Tissue Conc. (ug/Kg)**	**Remarks**
Aldrin	0.13	2,300	299	WQC was reduced by a factor of 10 to account for chronic effects; BCF estimate is based on Dieldrin since Aldrin rapidly transformed to Dieldrin in the environment; BCF is based on 1.1% lipid level for marine fish, Spot (*Leiostomus xanthurus*).
Dieldrin	0.0019	2,300	4.37	BCF is based on 1.1% lipid level for marine fish, Spot (*Leiostomus xanthurus*).
Total Chlordane	0.004	16,000	64	Total Chlordane includes alpha-chlordane, trans nonachlor, heptachlor, heptachlor epoxide; WQC for Chlordane is used for Total Chlordane; BCF is based on 3.6% lipid level for sheepshead minnow (*Cyprinodon variegatus*).
Total Endosulfan	0.0087	328	2.85	Total Endosulfan includes endosulfan I, endosulfan II and endosulfan sulfate; WQC for Endosulfan is used for Total Endosulfan; BCF is based on 3.6% lipid level for sheepshead minnow (*Cyprinodon variegatus*).

1,4-Dichlorobenzene	197	60	11820	Ambient conc. is based on lowest observed effect level (LOEL) for saltwater species from WQC, and reduced by a factor of 10 to account for chronic effects; BCF is based on the whole body for bluegill (*Lepomis macrochirus*).
Arsenic	36	350	12600	Ambient conc. is based on the saltwater criteria continuous conc. for arsenic (III); BCF is based on the Eastern Oyster (*Crassostrea virginica*).
Chromium	50	236	11800	Ambient conc. is based on Chromium (VI) since it is substantially more toxic than Chromium (III); BCF is based on polychaete worm.
Copper	2.9	3,300	9570	WQC is based on a hardness value of 100; BCF is based on soft shell clam.
Lead	8.5	1,400	11900	Ambient conc. is based on saltwater criteria continuous conc.; BCF is based on the Eastern Oyster (*Crassostrea virginica*).
Nickel	8.3	458	3802	Ambient conc. is based on saltwater criteria continuous conc.; BCF is based on the Eastern Oyster (*Crassostrea virginica*).

Silver	0.23	6,500	1495	Water Quality Criterion (WQC) was reduced by a factor of 10 to account for chronic effects; BCF is based on the Blue Mussel (*Mytilus edulis*).
Zinc	86	17,640	1517040	Ambient conc. is based on saltwater criteria continuous conc.; BCF is based on Eastern Oyster (*Crassostrea virginica*).

Compound	Ambient Conc. (ug/L)[1]	BCF	Tissue Conc. (ug/Kg)	Remarks
Aldrin	0.13	2,300	299	WQC was reduced by a factor of 10 to account for chroni based on Dieldrin since Aldrin rapidly transformed to Die BCF is based on 1.1% lipid level for marine fish, Spot (*L.*
Dieldrin	0.0019	2,300	4.37	BCF is based on 1.1% lipid level for marine fish, Spot (*L.*
Total Chlordane	0.004	16,000	64	Total Chlordane includes alpha-chlordane, trans nonachlo epoxide; WQC for Chlordane is used for Total Chlordane lipid level for sheepshead minnow (*Cyprinodon variegati*
Total Endosulfan	0.0087	328	2.85	Total Endosulfan includes endosulfan I, endosulfan II and for Endosulfan is used for Total Endosulfan; BCF is base sheepshead minnow (*Cyprinodon variegatus*).
1,4-Dichlorobenzene	197	60	11820	Ambient conc. is based on lowest observed effect level (I species from WQC, and reduced by a factor of 10 to acco BCF is based on the whole body for bluegill (*Lepomis ma*
Arsenic	36	350	12600	Ambient conc. is based on the saltwater criteria continuo BCF is based on the Eastern Oyster (*Crassostrea virginic*

Chromium	50	236	11800	Ambient conc. is based on Chromium (VI) since it is substantially more toxic than Chromium (III); BCF is based on polychaete worm.
Copper	2.9	3,300	9570	WQC is based on a hardness value of 100; BCF is based on soft shell clam.
Lead	8.5	1,400	11900	Ambient conc. is based on saltwater criteria continuous conc.; BCF is based on the Eastern Oyster (*Crassostrea virginica*).
Nickel	8.3	458	3802	Ambient conc. is based on saltwater criteria continuous conc.; BCF is based on the Eastern Oyster (*Crassostrea virginica*).
Silver	0.23	6,500	1495	Water Quality Criterion (WQC) was reduced by a factor of 10 to account for chronic effects; BCF is based on the Blue Mussel (*Mytilus edulis*).
Zinc	86	17,640	1517040	Ambient conc. is based on saltwater criteria continuous conc.; BCF is based on Eastern Oyster (*Crassostrea virginica*).

USEPA. 1980b. Ambient Water Quality Criteria for Aldrin/Dieldrin; EPA 440/5-80-019; December 1980.
USEPA. 1980c. Ambient Water Quality Criteria for Chlordane; EPA 440/5-80-027; October 1980.
USEPA. 1980d. Ambient Water Quality Criteria for Heptachlor; EPA 440/5-80-052; October 1980.
USEPA. 1980e. Ambient Water Quality Criteria for Endosulfan; EPA 440/5-80-046; October 1980.
USEPA. 1980f. Ambient Water Quality Criteria for Dichlorobenzenes; EPA 440/5-80-039; October 1980.
USEPA. 1984a. Ambient Water Quality Criteria for Lead - 1984; EPA 440/5-84-027; January 1985.
USEPA. 1984b. Ambient Water Quality Criteria for Copper - 1984; EPA 440/5-84-031; January 1985.
USEPA. 1985a. Ambient Water Quality Criteria for Chromium - 1984; EPA 440/5-84-029; January 1985.
USEPA. 1985c. Ambient Water Quality Criteria for Arsenic - 1984; EPA 440/5-84-033; January 1985.
USEPA. 1986. Ambient Water Quality Criteria for Nickel - 1986; EPA 440/5-86-004; September 1986.
USEPA. 1987b. Ambient Water Quality Criteria for Zinc - 1987; EPA 440/5-87-003.
USEPA. 1992a. Draft Ambient Water Quality Criteria for Silver.

ATTACHMENT B

Benthic Cancer Protection Level Calculations for the Protection of Human Health from the Consumption of Fish Exposed to Dredged Material at the Historic Area Remediation Site

Basis[fn1]:

10^{-4} Benthic Tissue Level (ug/kg) = $\frac{[10^{-4} \text{ Conc. in Fish }] \times [\text{Whole Body/fillet Factor (1.35)}]^{fn5}}{\text{Trophic Transfer Factor}^{fn2}}$

10^{-4} Conc. in Fish (ug/kg) = $\frac{\text{Toxicological Dose (ug/day)}}{[\text{Seafood Consumption (6.5 g/day)}^{fn3}] \times [10^{-3}\text{kg/g}]}$

Toxicological Dose (ug/day) = $\frac{[\text{Risk Level } (10^{-4})] \times [\text{Body Weight (70 kg)}^{fn3}] \times [10^{3} \text{ ug/mg}]}{\text{Potency Factor, } q_1^{*} \text{ (kg-day/mg)}^{fn4}}$

	Cancer Potency Factor q_1^* (kg-day/mg)	**Acceptable Concentration in Fish (ug/kg)**	**Trophic Transfer Factor**	**Benthic Protection Level (ug/kg)**
Pesticides				
Aldrin	17	63	2.6	33
Chlordane	1.3	828	2.3	486
Dieldrin	16	67	1.4	65
Heptachlor	4.5	239	2.7	120
Heptachlor epoxide	9.1	118	1.4	114
Industrial Organics				
1.4-Dichlorobenzene	0.024	44,872	1	60,577
PAHs				
Benzo(a)pyrene	7.3	147	0.1	2,000
METALS				
Arsenic	1.5	718	3	323

ATTACHMENT C

Benthic Non-Cancer Protection Level Calculations for the Protection of Human Health from the Consumption of Fish Exposed to Dredged Material at the Historic Area Remediation Site

Basis[fn1]:

$$\text{Benthic Tissue Level (ug/kg)} = \frac{[\text{Conc. in Fish}] \times [\text{Whole Body/fillet Factor (1.35)}]^{fn5}}{\text{Trophic Transfer Factor}^{fn2}}$$

$$\text{Conc. in Fish (ug/kg)} = \frac{\text{Toxicological Dose (ug/day)}}{[\text{Seafood Consumption (6.5 g/day)}^{fn3}] \times [10^{-3}\text{kg/g}]}$$

$$\text{Toxicological Dose (ug/day)} = [\text{Reference dose}^{fn4}] \times [\text{Body Weight (70 kg)}^{fn3}]$$

	Reference Dose RfD (ug/kg-day)	Acceptable Concentration in Seafood (ug/kg)	Trophic Transfer Factor	Benthic Protection Level (ug/kg)
Metals				
Arsenic	0.3	3,231	3	1,454
Chromium	5	54,000	1	73,000
Copper	37.1	400,000	1	540,000
Nickel	20	215,000	1	290,000
Silver	5	54,000	1	73,000
Zinc	300	3,230,769	1	4,361,538
Pesticides				
Aldrin	0.03	323	2.6	167
Chlordane	0.06	592	2.3	350
Dieldrin	0.05	538	1.4	518
Endosulfan	6	64,615	1	87,231
Heptachlor	0.5	5,385	2.7	2,692
Heptachlor epoxide	0.013	140	1.4	135
PAHs				
Acenaphthene	60	650,000	0.1	8,775,000
Fluorene	40	430,000	0.1	5,805,000
Phenanthrene	300	3,230,000	0.1	43,605,000
Anthracene	300	3,230,000	0.1	43,605,000
Fluoranthene	40	430,000	0.1	5,805,000
Pyrene	30	325,000	0.1	4,387,000

NOTES:

[fn1] Human health cancer and non-cancer assessments adapted from *Guidance for Assessing Chemical Contaminant Data for use in Fish Advisories: Volume II: Risk Assessment and Fish Consumption Limits.* U.S. Environmental Protection Agency, EPA823-B-94-004, Office of Science and Technology, Washington, DC, June 1994.

[fn2] Trophic transfer factors were calculated by Mr. Lawrence Burkhard, EPA Duluth, using the food chain transfer model developed by Gobas (1993).

[fn3] Default values were taken from EPA's national toxics rule for setting water quality criteria, USEPA (1992b).

[fn4] Cancer potency factors and non-cancer reference doses are taken from USEPA (1995).

[fn5] The acceptable concentration in seafood is defined on the basis of the fillet or edible portion for humans. Trophic transfer, however, was defined on the basis of whole body characteristics, including lipid concentrations. Experience in New York State indicates a whole body to fillet ration ranging from 1.2 to 1.5 is applicable to lipophilic substances. The mid range value of 1.35 is used in this analysis.

AH09-1 Compsite

MODEL: SHORT-TERM FATE OF DREDGED MATERIAL FROM SPLIT HULL BARGE OR HOPPER DREDGE
(PC Version 5.01 MAY, 1993)
(Extended Memory Modification: December, 1997)
This Version Supports Grid Sizes up to 96 x 96 Points

TITLE: AH09-1 Compsite

FILE: TmpFile .DUE

AREA: THE PROJECT AREA IS DESCRIBED BY A 96 X 96 GRID.

THERE ARE 96 GRID POINTS (NMAX) IN THE Z-DIRECTION (FROM LEFT TO RIGHT)
AND 96 GRID POINTS (MMAX) IN THE X-DIRECTION (FROM TOP TO BOTTOM).

SITE: THE DISPOSAL SITE IS REPRESENTED AS A RECTANGLE ON THE SITE GRID.

THE TOPMOST BOUNDARY IS LOCATED AT POINT #34 (MDS1) FROM THE TOP OF THE GRID.

THE BOTTOMMOST BOUNDARY IS LOCATED AT POINT #64 (MDS2) FROM THE TOP OF THE GRID.

THE LEFTMOST BOUNDARY IS LOCATED AT POINT #65 (NDS1) FROM THE LEFT OF THE GRID.

THE RIGHTMOST BOUNDARY IS LOCATED AT POINT #94 (NDS2) FROM THE LEFT OF THE GRID.

EXECUTION PARAMETERS:

MODEL COEFFICIENTS SPECIFIED IN INPUT DATA (KEY1 = 1).

VERTICAL DIFFUSION COEFFICIENT (AKY0) COMPUTED FROM PRITCHARD EQUATION (IPRIT = 1).

PERFORM COMPLETE ANALYSIS INCLUDING DESCENT, COLLAPSE, AND TRANSPORT-DIFFUSION (KEY2 = 0).

PERFORM TIER III OCEAN DUMPING INITIAL MIXING EVALUATION
TO COMPARE WITH TOXICITY CRITERIA (KEY3 = 3).

PRINTING OF CONVECTIVE DESCENT RESULTS REQUESTED (IPCN = 1).

PRINTING OF CONVECTIVE DESCENT RESULTS REQUESTED (IPCN = 1).

PRINTING OF DYNAMIC COLLAPSE RESULTS REQUESTED (IPCL = 1).

QUARTERLY PRINTING OF LONG-TERM TRANSPORT DIFFUSION RESULTS REQUESTED (IPLT = 0).

LONG-TERM TRANSPORT DIFFUSION RESULTS REQUESTED AT THE FOLLOWING 3 DEPTH(S):
0.00 FT
160.00 FT
320.00 FT

GRID: NUMBER OF LONG TERM GRID POINTS IN Z-DIRECTION (NMAX) = 96

NUMBER OF LONG TERM GRID POINTS IN X-DIRECTION (MMAX) = 96

GRID SPACING IN Z-DIRECTION (DZ) = 200.00000 FT

GRID SPACING IN X-DIRECTION (DX) = 200.00000 FT

CONSTANT DEPTH GRID SPECIFIED HAVING A DEPTH (DEPC) OF 330.00000 FT.

TIME (HR)	DEPTH (FT)	MAX CONC ABOVE BACKGROUND ON ENTIRE GRID (MG/L)	X-LOC (FT)	Z-LOC (FT)	MAX CONC ABOVE BACKGROUND OUTSIDE DISPOSAL SITE (MG/L)
1.00	0.0	0.671E+00	9400.	12000.	0.671E+00
2.00	0.0	0.472E-04	9400.	8400.	0.472E-04
3.00	0.0	0.581E-19	9400.	4800.	0.581E-19
4.00	0.0	0.681E-37	7600.	200.	0.681E-37
1.00	160.0	0.580E+02	9400.	12000.	0.580E+02
2.00	160.0	0.592E+01	9400.	8400.	0.592E+01
3.00	160.0	0.313E-02	9400.	4800.	0.313E-02
4.00	160.0	0.681E-37	7600.	200.	0.681E-37
1.00	320.0	0.949E+02	9400.	11800.	0.949E+02
2.00	320.0	0.122E+02	9400.	8400.	0.122E+02
3.00	320.0	0.299E+01	9400.	4800.	0.299E+01
4.00	320.0	0.297E+01	9400.	1200.	0.297E+01

INITIAL MIXING COMPUTATIONS RESULTS FOR CLAY :

TIME (HR)	DEPTH (FT)	MAX CONC ABOVE BACKGROUND ON ENTIRE GRID (MG/L)	X-LOC (FT)	Z-LOC (FT)	MAX CONC ABOVE BACKGROUND OUTSIDE DISPOSAL SITE (MG/L)
1.00	0.0	0.445E+01	9400.	12000.	0.445E+01
2.00	0.0	0.183E+01	9400.	8400.	0.183E+01
3.00	0.0	0.823E+00	9400.	4800.	0.823E+00
4.00	0.0	0.423E+00	9400.	1200.	0.423E+00
1.00	160.0	0.317E+02	9400.	12000.	0.317E+02
2.00	160.0	0.119E+02	9400.	8400.	0.119E+02
3.00	160.0	0.498E+01	9400.	4800.	0.498E+01
4.00	160.0	0.246E+01	9400.	1200.	0.246E+01
1.00	320.0	0.311E+02	9400.	11800.	0.311E+02
2.00	320.0	0.157E+02	9400.	8200.	0.157E+02
3.00	320.0	0.847E+01	9400.	4600.	0.847E+01
4.00	320.0	0.120E+02	9400.	1000.	0.120E+02

INITIAL MIXING COMPUTATIONS RESULTS FOR Silt :

TIME (HR)	DEPTH (FT)	MAX CONC ABOVE BACKGROUND ON ENTIRE GRID (MG/L)	X-LOC (FT)	Z-LOC (FT)	MAX CONC ABOVE BACKGROUND OUTSIDE DISPOSAL SITE (MG/L)
1.00	0.0	0.121E+01	9400.	12000.	0.121E+01
2.00	0.0	0.847E-01	9400.	8400.	0.847E-01
3.00	0.0	0.213E-02	9400.	4800.	0.213E-02
4.00	0.0	0.926E-05	9400.	1200.	0.926E-05

1.00	160.0	0.224E+02	9400.	12000.	0.224E+02
2.00	160.0	0.615E+01	9400.	8400.	0.615E+01
3.00	160.0	0.172E+01	9400.	4800.	0.172E+01
4.00	160.0	0.405E+00	9400.	1200.	0.405E+00
1.00	320.0	0.333E+02	9400.	11800.	0.333E+02
2.00	320.0	0.133E+02	9400.	8200.	0.133E+02
3.00	320.0	0.399E+01	9400.	4600.	0.399E+01
4.00	320.0	0.686E+00	9400.	1200.	0.686E+00

INITIAL MIXING COMPUTATIONS RESULTS FOR FLUID :

TIME (HR)	DEPTH (FT)	MAX CONC ABOVE BACKGROUND ON ENTIRE GRID (PERCENT)	X-LOC (FT)	Z-LOC (FT)	MAX CONC ABOVE BACKGROUND OUTSIDE DISPOSAL SITE (PERCENT)
0.17	0.0	0.340E-37	8200.	13600.	0.000E+00
0.33	0.0	0.272E-37	8200.	13000.	0.000E+00
0.50	0.0	0.221E-37	8200.	12400.	0.221E-37
0.67	0.0	0.181E-37	8200.	11800.	0.181E-37
0.83	0.0	0.150E-37	8000.	11000.	0.150E-37
1.00	0.0	0.125E-37	8000.	10400.	0.125E-37
1.17	0.0	0.105E-37	8000.	9800.	0.105E-37
1.33	0.0	0.888E-38	8000.	9200.	0.888E-38
1.50	0.0	0.757E-38	7800.	8400.	0.757E-38
1.67	0.0	0.650E-38	7800.	7800.	0.650E-38
1.83	0.0	0.561E-38	7800.	7200.	0.561E-38
2.00	0.0	0.486E-38	7600.	6400.	0.486E-38
2.17	0.0	0.424E-38	7600.	5800.	0.424E-38
2.33	0.0	0.371E-38	7600.	5200.	0.371E-38
2.50	0.0	0.326E-38	7600.	4400.	0.326E-38
2.67	0.0	0.288E-38	7400.	3800.	0.288E-38
2.83	0.0	0.255E-38	7400.	3200.	0.255E-38
3.00	0.0	0.227E-38	7200.	2400.	0.227E-38
3.17	0.0	0.203E-38	7200.	1800.	0.203E-38
3.33	0.0	0.181E-38	7200.	1200.	0.181E-38
3.50	0.0	0.163E-38	7000.	400.	0.163E-38
3.67	0.0	0.147E-38	7000.	200.	0.147E-38
3.83	0.0	0.132E-38	7000.	200.	0.132E-38
4.00	0.0	0.120E-38	6800.	200.	0.120E-38
0.17	160.0	0.340E-37	8200.	13600.	0.000E+00
0.33	160.0	0.272E-37	8200.	13000.	0.000E+00
0.50	160.0	0.221E-37	8200.	12400.	0.221E-37
0.67	160.0	0.181E-37	8200.	11800.	0.181E-37
0.83	160.0	0.150E-37	8000.	11000.	0.150E-37
1.00	160.0	0.125E-37	8000.	10400.	0.125E-37
1.17	160.0	0.105E-37	8000.	9800.	0.105E-37
1.33	160.0	0.888E-38	8000.	9200.	0.888E-38
1.50	160.0	0.757E-38	7800.	8400.	0.757E-38
1.67	160.0	0.650E-38	7800.	7800.	0.650E-38
1.83	160.0	0.561E-38	7800.	7200.	0.561E-38
2.00	160.0	0.486E-38	7600.	6400.	0.486E-38
2.17	160.0	0.424E-38	7600.	5800.	0.424E-38
2.33	160.0	0.371E-38	7600.	5200.	0.371E-38
2.50	160.0	0.326E-38	7600.	4400.	0.326E-38
2.67	160.0	0.288E-38	7400.	3800.	0.288E-38
2.83	160.0	0.255E-38	7400.	3200.	0.255E-38
3.00	160.0	0.227E-38	7200.	2400.	0.227E-38
3.17	160.0	0.203E-38	7200.	1800.	0.203E-38
3.33	160.0	0.181E-38	7200.	1200.	0.181E-38

3.50	160.0	0.163E-38	7000.	400.	0.163E-38
3.67	160.0	0.147E-38	7000.	200.	0.147E-38
3.83	160.0	0.132E-38	7000.	200.	0.132E-38
4.00	160.0	0.120E-38	6800.	200.	0.120E-38

0.17	320.0	0.105E+01	9400.	14800.	0.000E+00
0.33	320.0	0.950E+00	9400.	14200.	0.000E+00
0.50	320.0	0.851E+00	9400.	13600.	0.116E-01
0.67	320.0	0.755E+00	9400.	13000.	0.533E+00
0.83	320.0	0.666E+00	9400.	12400.	0.666E+00
1.00	320.0	0.585E+00	9400.	11800.	0.585E+00
1.17	320.0	0.512E+00	9400.	11200.	0.512E+00
1.33	320.0	0.447E+00	9400.	10600.	0.447E+00
1.50	320.0	0.391E+00	9400.	10000.	0.391E+00
1.67	320.0	0.341E+00	9400.	9400.	0.341E+00
1.83	320.0	0.298E+00	9400.	8800.	0.298E+00
2.00	320.0	0.260E+00	9400.	8200.	0.260E+00
2.17	320.0	0.228E+00	9400.	7600.	0.228E+00
2.33	320.0	0.199E+00	9400.	7000.	0.199E+00
2.50	320.0	0.175E+00	9400.	6400.	0.175E+00
2.67	320.0	0.154E+00	9400.	5800.	0.154E+00
2.83	320.0	0.135E+00	9400.	5200.	0.135E+00
3.00	320.0	0.119E+00	9400.	4600.	0.119E+00
3.17	320.0	0.105E+00	9400.	4000.	0.105E+00
3.33	320.0	0.933E-01	9400.	3400.	0.933E-01
3.50	320.0	0.828E-01	9400.	2800.	0.828E-01
3.67	320.0	0.736E-01	9400.	2200.	0.736E-01
3.83	320.0	0.655E-01	9400.	1600.	0.655E-01
4.00	320.0	0.585E-01	9400.	1000.	0.585E-01

0.17	316.9	0.163E+01	9400.	14800.	0.000E+00
0.33	316.9	0.134E+01	9400.	14200.	0.000E+00
0.50	316.9	0.111E+01	9400.	13600.	0.150E-01
0.67	316.9	0.918E+00	9400.	13000.	0.648E+00
0.83	316.9	0.768E+00	9400.	12400.	0.768E+00
1.00	316.9	0.647E+00	9400.	11800.	0.647E+00
1.17	316.9	0.548E+00	9400.	11200.	0.548E+00
1.33	316.9	0.467E+00	9400.	10600.	0.467E+00
1.50	316.9	0.401E+00	9400.	10000.	0.401E+00
1.67	316.9	0.345E+00	9400.	9400.	0.345E+00
1.83	316.9	0.299E+00	9400.	8800.	0.299E+00
2.00	316.9	0.260E+00	9400.	8200.	0.260E+00
2.17	316.9	0.228E+00	9400.	7600.	0.228E+00
2.33	316.9	0.200E+00	9400.	7000.	0.200E+00
2.50	316.9	0.176E+00	9400.	6400.	0.176E+00
2.67	316.9	0.156E+00	9400.	5800.	0.156E+00
2.83	316.9	0.138E+00	9400.	5200.	0.138E+00
3.00	316.9	0.123E+00	9400.	4600.	0.123E+00
3.17	316.9	0.110E+00	9400.	4000.	0.110E+00
3.33	316.9	0.986E-01	9400.	3400.	0.986E-01
3.50	316.9	0.887E-01	9400.	2800.	0.887E-01
3.67	316.9	0.799E-01	9400.	2200.	0.799E-01
3.83	316.9	0.722E-01	9400.	1600.	0.722E-01
4.00	316.9	0.654E-01	9400.	1000.	0.654E-01

RESULT: THE TOXICITY CRITERIA FOR THE DISPOSAL SITE WAS NOT VIOLATED.

*** RUN COMPLETED ***

Paul Berman

From: Cortes, Javier SAJ [Javier.Cortes@usace.army.mil]
Sent: Wednesday, January 13, 2010 2:21 PM
To: Pberman@ANAMARinc.com
Subject: FW: FW: Arecibo AEC:0010426
Attachments: ADDAMS Model inputs from COE and EPA.doc

Categories: [CRM] Regarding: Arecibo Harbor 103 - 2009

Paul

Here is your answer.

Thanks

Javier

-----Original Message-----
From: Reiss.Mark@epamail.epa.gov [mailto:Reiss.Mark@epamail.epa.gov]
Sent: Wednesday, January 13, 2010 1:59 PM
To: Cortes, Javier SAJ
Subject: Re: FW: Arecibo AEC:0010426

Use the same input file as San Juan except set the water depth to constant water depth to 330 feet...and make sure the bottom of the density profile of the water column does not exceed that depth

"Cortes, Javier SAJ" <Javier.Cortes@usace.army.mil>
01/12/2010 10:37 AM

To: Mark Reiss/R2/USEPA/US@EPA
cc:
Subject: FW: Arecibo AEC:0010426

Hi Mark,

How things are doing? We are having a cool winter here in Florida.

We need help with the ADDAMS model inputs for Arecibo Harbor.

Thanks

Javier

-----Original Message-----
From: Paul Berman [mailto:Pberman@ANAMARinc.com]
Sent: Friday, January 08, 2010 5:09 PM
To: Cortes, Javier SAJ
Subject: RE: Arecibo AEC:0010426

I created a table with the inputs needed to run the simulation.

Paul Berman
QAQC Officer
ANAMAR Environmental Consulting
Phone: 352-377-5770

-----Original Message-----
From: Cortes, Javier SAJ [mailto:Javier.Cortes@usace.army.mil]
Sent: Friday, January 08, 2010 4:58 PM
To: Paul Berman
Subject: RE: Arecibo

Paul,

What type of inputs?

Thanks

Javier

-----Original Message-----
From: Paul Berman [mailto:Pberman@ANAMARinc.com]
Sent: Friday, January 08, 2010 4:45 PM
To: Cortes, Javier SAJ
Subject: Arecibo

Javier,

I am going to need the inputs for the Arecibo ODMDS for the ADDAMS model for the report. Can you send them to me sometime in the next week or two?

Thanks,

Paul Berman

QAQC Officer

ANAMAR Environmental Consulting

Phone: 352-377-5770

(See attached file: ADDAMS Model inputs from COE and EPA.doc)